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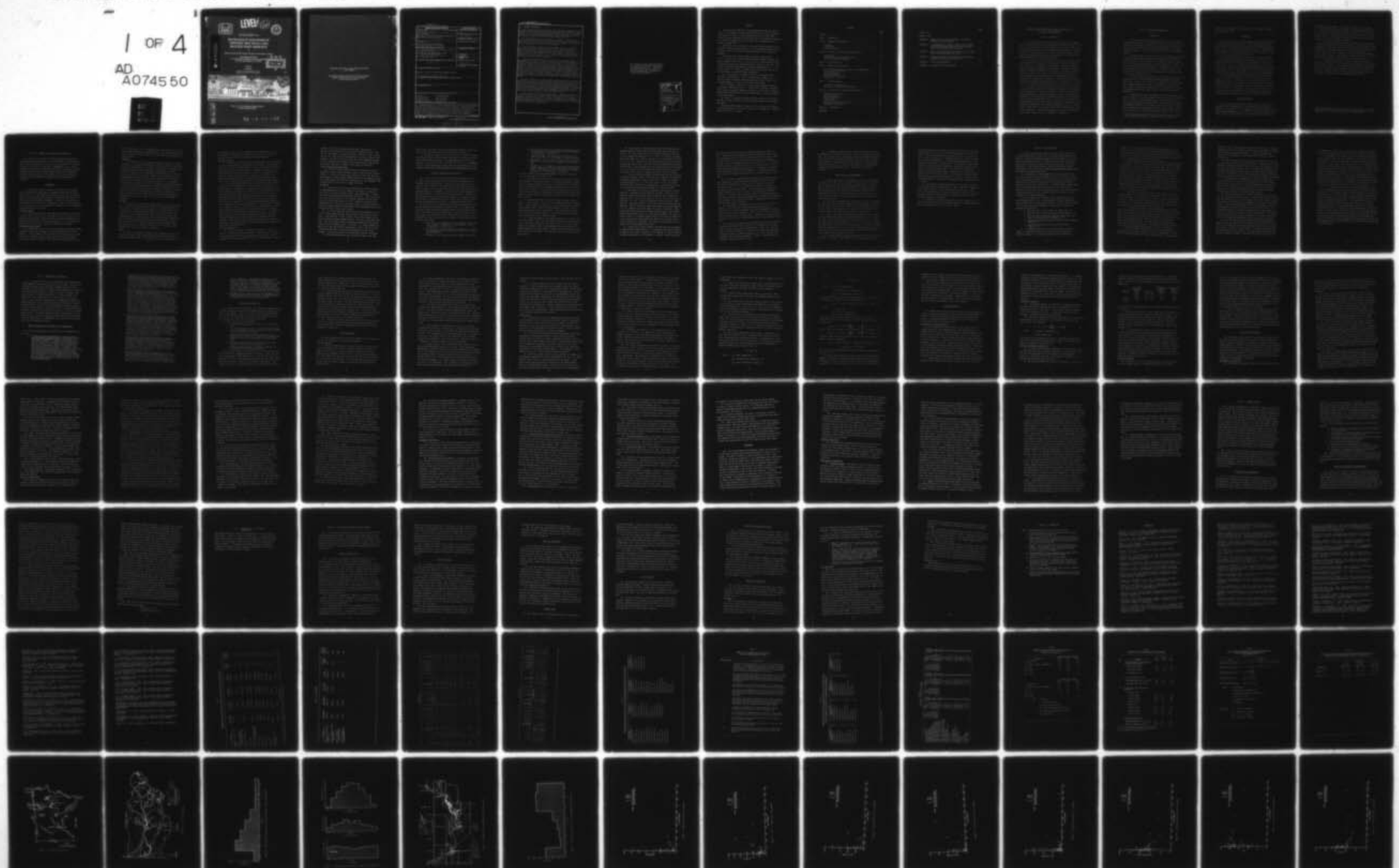

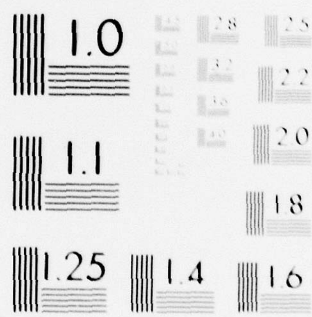

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TECHNICAL REPORT EL-79-5

WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE WILD RICE RIVER, MINNESOTA

by

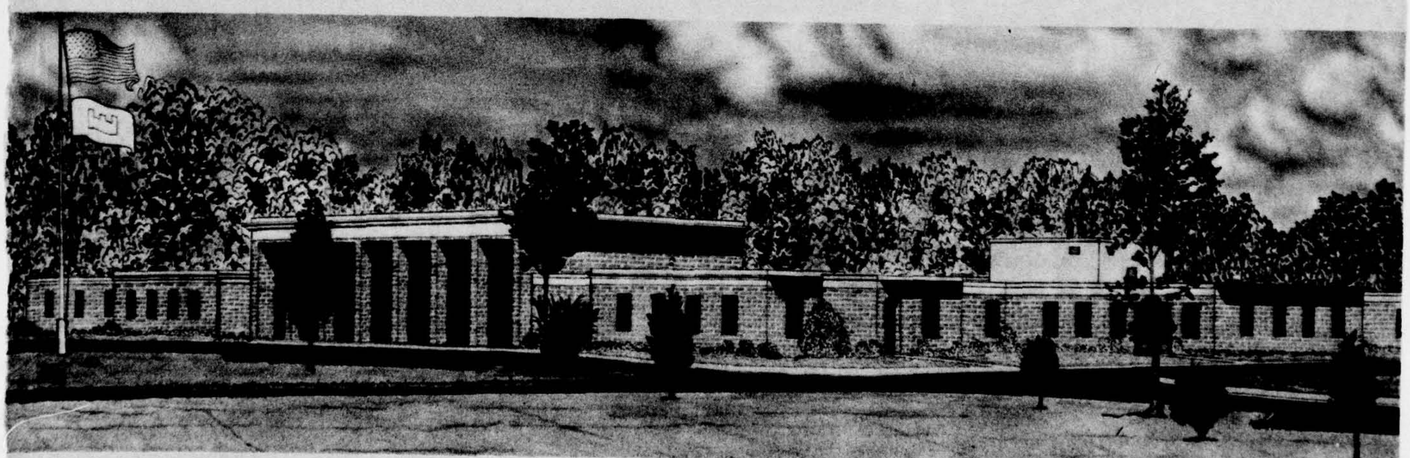
Dennis E. Ford, Kent W. Thornton, W. Bryan Ford III, Allan S. Lessem

Environmental Laboratory
U. S. Army Engineer Waterways Experiment Station
P. O. Box 631, Vicksburg, Miss. 39180

July 1979

Final Report

Approved For Public Release; Distribution Unlimited



Prepared for U. S. Army Engineer District, St. Paul
St. Paul, Minnesota 55101

Unclassified

SECURITY CLASSIFICATION OF THIS PAGE (When Data Entered)

REPORT DOCUMENTATION PAGE		READ INSTRUCTIONS BEFORE COMPLETING FORM
1. REPORT NUMBER Technical Report EL-79-5	2. GOVT ACCESSION NO.	3. RECIPIENT'S CATALOG NUMBER
4. TITLE (and Subtitle) WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE, WILD RICE RIVER, MINNESOTA		5. TYPE OF REPORT & PERIOD COVERED Final report
		6. PERFORMING ORG. REPORT NUMBER
7. AUTHOR(s) Dennis E. Ford, Kent W. Thornton, W. Bryan Ford III, Allan S. Lessem		8. CONTRACT OR GRANT NUMBER(s)
9. PERFORMING ORGANIZATION NAME AND ADDRESS U. S. Army Engineer Waterways Experiment Station Environmental Laboratory P. O. Box 631, Vicksburg, Miss. 39180		10. PROGRAM ELEMENT, PROJECT, TASK AREA & WORK UNIT NUMBERS
11. CONTROLLING OFFICE NAME AND ADDRESS U. S. Army Engineer District, St. Paul St. Paul, Minn. 55101		12. REPORT DATE July 1979
		13. NUMBER OF PAGES 292
14. MONITORING AGENCY NAME & ADDRESS(if different from Controlling Office) (12) 299		15. SECURITY CLASS. (of this report) Unclassified
		15a. DECLASSIFICATION/DOWNGRADING SCHEDULE

16. DISTRIBUTION STATEMENT (of this Report)

Approved for public release; distribution unlimited.

(14) WES-EL-79-5

17. DISTRIBUTION STATEMENT (of the abstract entered in Block 20, if different from Report)

18. SUPPLEMENTARY NOTES

19. KEY WORDS (Continue on reverse side if necessary and identify by block number)

Bioassays	Nutrient loadings
Ecological models	Twin Valley Lake
Eutrophication	Water Quality
Mathematical models	Wild Rice River

20. ABSTRACT (Continue on reverse side if necessary and identify by block number)

The water quality of proposed Twin Valley Lake, Wild Rice River, Minnesota, was evaluated with respect to its eutrophication potential and the water quality criteria and standards appropriate for project purposes. Study approaches included review of existing water quality data taken on the Wild Rice River, review of existing data on surrounding lakes, algal bioassay analyses on water samples taken in the Wild Rice River and in one nearby lake, laboratory studies on soil samples taken at the proposed project site, mathematical simulations, and nutrient loading analyses. (Continued)

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Unclassified

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20. ABSTRACT (Continued).

Water quality data from the Wild Rice River indicated no major water quality problems. Dissolved oxygen averaged 94 percent saturation. Phosphorus concentrations were sufficiently high to support nuisance algal blooms, but nitrogen concentrations were low, indicating possible nitrogen limitation. Fecal coliform counts exceeded 200 colonies/100 ml only during storm events.

The surrounding lakes can be classified as eutrophic. Lakes that were morphometrically similar to proposed Twin Valley Lake did not stratify and were able to meet the State standard of 5 mg/ℓ for dissolved oxygen. The algae concentrations ranged from 1 to 130 μg/ℓ chlorophyll a.

Algal bioassays were conducted on water samples taken in the Wild Rice River, Dayton Hollow Reservoir, and Ottertail River, the major tributary to Dayton Hollow Reservoir. Chemical analyses on the samples taken in the Ottertail River and Dayton Hollow Reservoir indicated that conditions in the reservoir and river were similar. In all of the samples, the nutrients were in an available form. The bioassays were inconclusive, indicating that either phosphorus or nitrogen could be limiting.

A reservoir ecological model was used to predict the trophic status of the impoundment and the effects of various reservoir operational and management schemes on the water quality. Monte Carlo simulations were used to include the variability inherent in the update data and coefficients. The mathematical simulations indicated that the downstream natural temperature objective could be met with either selective or bottom withdrawal. Bottom withdrawal is recommended. The lake will probably stratify intermittently from May through June, but State dissolved oxygen standards will not be violated. Simulated algal concentrations were 2 to 90 μg/ℓ chlorophyll a. Blue-green algae were predicted to dominate throughout the summer. No problems with fecal coliforms exceeding standards were predicted, but intermittent problems in the headwater regions are probable.

Laboratory studies on soil samples taken at the project site were used to study soil-water interactions under anaerobic conditions and to provide guidance on reservoir clearing and filling. The studies indicated that anoxia would develop 5 to 15 days after stratification and that hydrogen sulfide production is possible in another 5 to 15 days. Prior to filling, removal of all vegetation from the reservoir would probably reduce the initial oxygen demand. A series of fillings and flushing followed by a slow incremental filling would probably improve water quality during the initial years after filling.

Appendix A describes the algal assay procedures. Appendix B describes the laboratory experiments on soil samples taken at the proposed project site. Appendix C discusses the potential for aquatic macrophyte growth. Appendix D presents the initial conditions, coefficients, and updates for the mathematical ecological simulations. Appendix E lists the coefficient references. Appendix F presents the nutrient loading results.

PREFACE

This study was conducted by the Environmental Laboratory (EL) of the U. S. Army Engineer Waterways Experiment Station (WES), Vicksburg, Mississippi, for the U. S. Army Engineer District, St. Paul (NCS). The project was authorized by Intra-Army Order for Reimbursable Services No. NCS 1A-77-127-EDH dated 19 September 1977 and Change Order No. 1 dated 5 December 1977.

This report is an evaluation of the water quality expected in the proposed Twin Valley Lake relative to its eutrophication potential and to water quality criteria and standards appropriate for the project purposes.

The research was conducted under the direct supervision of Mr. D. L. Robey, Chief, Ecosystem Modeling Branch (EMB), and under the general supervision of Drs. R. L. Eley, Chief, Ecosystem Research and Simulation Division, EL, and John Harrison, Chief, EL. Drs. D. E. Ford and K. W. Thornton, EMB, served as principal investigators. Dr. A. S. Lessem and Mr. W. B. Ford, III, EMB, participated in the study and in the development of the Monte Carlo simulations.

Ms. C. Stirgus, Ms. F. Thompson, and Mr. T. Quasebarth, EMB, assisted in data analysis. Drs. D. Gunnison and J. Barko, Ecosystem Processes Research Branch, prepared Appendixes B and C, respectively, and along with Mr. J. Norton, reviewed the draft report.

Drs. R. Megard, J. Shapiro, H. Stefan, and H. Wright, University of Minnesota, Minneapolis; Drs. T. Collins and D. Mathiason, Moorhead State College, Moorhead, Minnesota; and Drs. M. Bromel and J. Peterka, North Dakota State University, Fargo, North Dakota, provided assistance in data compilation.

Ms. Leslie A. Gardner, Utah Water Research Laboratory, Utah State University, Logan, conducted the algal bioassay analyses and prepared Appendix A.

Director of WES during the conduct of this study and the preparation and publication of this report was COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

CONTENTS

WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE, WILD RICE RIVER, MINNESOTA

PART I: INTRODUCTION

1. The proposed Twin Valley Lake would be formed by the impoundment of the Wild Rice River near Twin Valley, Minnesota (Figure 1). The project was authorized by the Flood Control Act of 1970 for purposes of flood control, recreation, and fish and wildlife development. The project would be operated primarily to reduce flooding downstream of Twin Valley on the Wild Rice River and on the Red River of the North from Halstad, Minnesota, to Drayton, North Dakota.

2. The objectives of this study were to evaluate the water quality and eutrophication potential of proposed Twin Valley Lake relative to project purposes. A combination of techniques including comparisons with surrounding impoundments, algal bioassays, laboratory studies on anaerobic conditions, mathematical simulations, and nutrient loading analyses was used. While each of the individual techniques have inherent assumptions and limitations, their combined use provides corroborative and complementary information. This approach has been used in other water quality studies (Thornton et al. 1976; Hall et al. 1977; Ford et al. 1978) and is described in detail by Thornton et al. (1977b).

3. This report presents results of a water quality evaluation of proposed Twin Valley Lake. The assumptions, limitations, and results of each technique are presented separately. Predictions from the different techniques are evaluated and compared based on the appropriate assumptions and limitations to determine the water quality and trophic status of proposed Twin Valley Lake. The specifics of the algal bioassay analyses, the laboratory studies on anaerobic conditions, and the macrophyte assessment are described in Appendixes A, B, and C, respectively. Model coefficients, updates, and initial conditions are specified in Appendix D. Coefficient references are listed in Appendix E, and nutrient loading results are included in Appendix F.

PART II: BACKGROUND INFORMATION

Watershed

4. The Wild Rice River watershed encompasses about 5130 km^2 in northwestern Minnesota. The Wild Rice River starts in Upper Rice Lake (elevation 458.1 m* msl) and flows in a generally westerly direction until it joins the Red River of the North 48 km north of Moorhead, Minnesota (Figure 2). The total length of the river is approximately 300 km.

5. Proposed Twin Valley Lake is located at approximately kilometre 98 near the physiographic transition zone from Red River lowland to glacial moraine. The topography of the 2300 km^2 of moraine located above (east) the proposed project is gently undulating to rugged. Land use in the watershed above the project is approximately 43 percent cultivated and 40 percent forested with 17 percent in pasture, old fields, marsh, bogs, and lakes. The percentage of forested lands increases in an easterly direction.

6. Point discharges of pollution to the Wild Rice River above the proposed project are the Mahnomen and Waubum wastewater treatment facilities. Both facilities consist of a pumping station and primary and secondary stabilization ponds. The Minnesota Pollution Control Agency (MPCA) considered the Waubum system to be adequate, but found the Mahnomen ponds reached capacity before stabilization occurred (MPCA 1975).

7. Numerous small lakes and swamps in the upper watershed store and delay runoff. The average annual runoff for 49 years of record at the U. S. Geological Survey (USGS) gage at Twin Valley is 6.5 cm. Generally, the streamflow at Twin Valley rises in late March or April from snowmelt (Figure 3). The largest flow usually occurs in April. The flow remains high through June and then slowly recedes. Approximately 62 percent of the total annual flow at Twin Valley occurs in

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* Elevations cited herein are in metres referred to mean sea level (msl); sampling elevations are referred to base of reservoir.

3 months--April through June. During periods of drought, the upper lakes sustain the flow.

Climatology

8. Weather observations of air temperature and precipitation are available from three National Weather Service (NWS) meteorological stations within the Wild Rice River basin at Ada, Mahnomen, and Twin Valley. The closest Class A meteorological station is approximately 65 km southwest of the proposed project in Fargo, North Dakota. Data from this station were used in this study.

9. The continental climate of the region is characterized by extreme variations in temperature and moderate precipitation. Mean annual precipitation and evaporation are approximately 53 and 76 cm, respectively. At Ada, the maximum recorded annual precipitation of 85 cm occurred in 1941, and the minimum recorded annual precipitation of 26 cm occurred in 1910. Approximately 70 percent of the annual precipitation occurs during the growing season of May through September.

10. At Fargo, North Dakota, the mean monthly air temperature ranges from $-13.8^{\circ}C$ in January to $21.8^{\circ}C$ in July (Figure 4a). The maximum mean monthly precipitation of 7.7 cm occurs in June while the minimum of 1.5 cm occurs in January and February (Figure 4b). Mean monthly wind speeds are relatively constant varying from a minimum of 5.4 m/sec in July to a maximum of 7.2 m/sec in April (Figure 4c). The prevailing wind is south-southeast for the months of June through October and north-northwest for the months of November through May.

Proposed Reservoir

11. The reservoir as discussed in this report is located at an alternative site approximately 1.5 km upstream from the original authorized site (Figure 5). The conservation pool (elevation 324 m msl) would impound approximately $9.25 \times 10^6 m^3$ of water with a surface area of 218 ha. This project would create a lake 9.5 m deep at the dam and

extending 11.3 km upstream. At full pool elevation of 336.5 m msl, $6.44 \times 10^7 m^3$ of water would be impounded, inundating approximately 684 ha. The maximum depth during full pool elevation is 22 m. The theoretical hydraulic residence time based on mean annual flow of 5 m^3/sec would be 21 days. Mean monthly residence times would vary from 7 days in April to 47 days in September (Figure 6).

12. The outlet works for proposed Twin Valley Lake would consist of a gated spillway and a selective withdrawal structure. The selective withdrawal structure would contain two water quality gates and two flood control gates. The primary functions of the structure would be to release normal river flows, temporarily stored flood waters, and emergency reservoir drawdown. The emergency spillway (elevation 336.5 m msl) would be used to pass all flow in excess of the design flood.

13. As currently envisioned, the project would be operated primarily for flood control with the operating plan enhancing other project purposes where practical. The normal pool level would be maintained at 324 m msl. Releases would be limited to 70 m^3/sec.* Above elevation 338.6 m msl releases would be uncontrolled and approximately equal to the inflow. Minimum flow would be approximately 0.14 m^3/sec.

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* This revised value differs from the 48 m^3/sec specified in the Design Memorandum (U. S. Army Engineer District, St. Paul 1975b).

PART III: SURVEY OF EXISTING WATER QUALITY DATA

14. There are two reasons for analyzing existing riverine and impoundment water quality data. First and foremost, extrapolating the data to the proposed project is one of the best approaches for predicting the water quality of the project. Impoundments in the same geographical area will have similar macrometeorology and may have similar watershed characteristics, thermal regimes, and biotic communities. Second, stream and lake data are required input to other predictive techniques (e.g., mathematical simulations and nutrient loading analyses).

Stream Data

15. Water quality data were taken by the USGS in the Wild Rice River at Twin Valley (USGS Gage No. 05062500) during 1975-1977 (USGS 1975, 1976, 1977). This sampling station was located near the proposed damsite (ca 2 km downstream), and the data should be representative of water entering the proposed impoundment. The data for the open water period March through October are summarized in Table 1 and in the form of scatter plots for selected parameters in Figures 7-15. All parameters varied extensively at low flow.

Dissolved oxygen

16. Dissolved oxygen (DO) concentrations averaged 94 percent saturation for the 3 years of data. The range was 65 to 115 percent. The minimum recorded DO of 6.1 mg/ℓ indicated no existing problems with DO concentrations satisfying criteria for aquatic life. No relationships between flow, time of year, and percent saturation were found.

Biochemical oxygen demand

17. During 1976 and 1977 the 5-day biochemical oxygen demand (BOD5) averaged 5.5 mg/ℓ with a maximum of 23 mg/ℓ recorded on 19 July 1977 (Table 1). The mean BOD5 concentration in 1977 was higher than in 1976 (8.5 versus 2.6 mg/ℓ), yet the percent saturation of DO was similar (94 percent versus 93 percent). Since low flows characterized the last

half of 1976 and all of 1977 through October and since there was little relationship between flow and BOD5 (Figure 7), the higher organic loadings upstream in 1977 may possibly be in the form of a point source of organic matter.

Phosphorus

18. Phosphorus is one of the major nutrients required for plant growth and it is usually considered to be the key nutrient in the acceleration of eutrophication. According to Wetzel (1975), uncontaminated waters generally have total phosphorus concentrations of 0.01 to 0.05 mg/ℓ. Sawyer's critical phosphorus concentration at spring overturn was 0.01 mg/ℓ (Sawyer 1954). Average total phosphorus (P) and orthophosphate (PO_4-P) concentrations for the period 1975 through 1977 were 0.057 and 0.024 mg/ℓ P, respectively (Table 1). The maximum P concentration recorded was 0.24 mg/ℓ. Phosphorus concentrations tended to increase with flow during major flood events (Figure 16), but the overall relationship with flow was not significant (Figure 8) because of the wide variance in concentrations at low flow. The mean ratio of PO_4-P to total P was 0.48. This ratio was in general agreement with Omernik (1977) and is an estimate of the fraction of bioavailable phosphorus.

Nitrogen

19. Total soluble inorganic nitrogen (TSIN = NO_3-N + NO_2-N + NH_4-N) is a measure of several available forms of nitrogen required for plant growth. The mean TSIN concentration of 0.11 mg/ℓ (Table 1) was relatively low and comparable with Sawyer's critical nitrogen concentration of 0.2 to 0.3 mg/ℓ (Sawyer 1954). Mean nitrate (NO_3-N) and ammonium (NH_4-N) concentrations were 0.066 and 0.041 mg/ℓ, respectively. All nitrite (NO_2-N) concentrations were less than detection limits. Concentrations of all forms of nitrogen were generally higher during the spring flood (e.g., Figures 17 and 18), but varied widely at low flow and from year to year (Figures 10 and 11).

Silica

20. Silica is required by diatoms in the synthesis of their frustule cell structure. When concentrations fall below 0.5 mg/ℓ, most diatoms cannot effectively compete with nonsiliceous algae (Wetzel 1975).

In the Wild Rice River, silica concentrations ranged from 1.5 to 22.0 mg/ℓ (Table 1). The mean concentration was 13.3 mg/ℓ. No problems were anticipated with silica limiting the diatom population.

Alkalinity and pH

21. The average alkalinity and pH in the Wild Rice River during the 3 years were 248.0 mg/ℓ as $CaCO_3$ and 8.2, respectively. The respective ranges were 150.0 to 347.0 mg/ℓ and 7.3 to 8.8. During 1976, pH varied periodically (Figure 19), possibly due to rainfall. The pH and CO_2 concentrations are inversely related; as rain falls through the atmosphere, it becomes saturated with CO_2, and since rain droplets are essentially unbuffered, the CO_2 concentrations depress the pH values. These cycles observed in pH during 1976 correspond with storm events.

22. The interaction of pH and CO_2 also influences other chemical constituents such as phosphorus. At pH 6 and above, progressively greater amounts of phosphate are associated with calcium (Wetzel 1975). Since phosphate concentrations are generally in the microgram per litre range and calcium and other major cation concentrations are in the milligrams per litre range, complex formations of these cations with phosphate anions will have little effect on the cation distribution, but will have a profound effect on the phosphate distribution (Wetzel 1975). Wetzel (1975) further stated that in a solution at pH 7 without other compounds, 40 mg/ℓ of calcium will limit the solubility of phosphate to about 10 μg/ℓ, whereas a calcium concentration of 100 mg/ℓ will limit the equilibrium value of phosphate to 1 μg/ℓ. In addition, Otsuki and Wetzel (1972) found that increasing pH leads to calcium carbonate formation and the coprecipitation of phosphate with the carbonate. The relatively high alkalinities and pH's may result in an appreciable concentration of $CaCO_3$ and coprecipitation of phosphorus in the proposed Twin Valley Lake.

Specific conductance

23. Specific conductance provides an estimate of a water's capacity to conduct an electrical current. This capacity is related to the concentrations of ionized substances in the water. Specific conductance can be measured readily in the field and provides a rough

estimate of the total dissolved solids (TDS) concentration. In general, TDS is 60 percent of the specific conductance value. This general relationship is apparent in Table 1 except for 1975. During the wet year, 1975, TDS was approximately 75 percent of specific conductance. The TDS values are all less than the recommended maximum value of 500 mg/ℓ for irrigation water but are greater than the EPA criterion of 250 mg/ℓ for domestic water supplies (EPA 1976).

Fecal coliforms and streptococci

24. Counts of fecal coliform and total streptococcus bacteria indicated no public health problems related to body contact recreation (Table 1). All fecal coliform counts except one were less than the Environmental Protection Agency (EPA) criteria and the Minnesota standard of 200 colonies per 100 ml for bathing waters. The maximum of 390 colonies/100 ml occurred after a summer rain storm.

Discussion

25. Care must be taken in extrapolating data from the stream to the impoundment because of the change in flow regime. As the water flows from a riverine environment into a lake environment, its depth and travel times increase and turbulence decreases. There is less surface aeration, more decay per unit distance traveled, and more opportunity for particulate matter to settle. Therefore, water quality consitutents that decay with time and/or adhere to particulate matter should be found in smaller concentrations in the impoundment than in the stream, provided no other source exists.

26. It is also important that the data be representative of a variety of flow regimes and that the data only be used within its confines. Time histories of flow for the 3 years of data illustrate different hydrologic regimes (Figures 20-22). Two major floods occurred in 1975. The first flood resulted from spring snowmelt while the second resulted from a midsummer storm. In contrast, the spring snowmelt was much less in 1976 and virtually nonexistent in 1977. Most of the data collected during these 3 years were representative of dry and wet extremes. Very little data were collected under "average" conditions.

27. DO concentrations were near saturation; however, the high

BOD5 in 1977 could pose a potential DO depletion problem, as decay per unit distance traveled increases in the proposed reservoir.

28. Phosphorus concentrations in the river were typical of nonpolluted waters while nitrogen concentrations were low. Concentrations of both nutrients should be lower in the impoundment than in the river, provided no other sources exist. The mean N/P ratio (i.e., TSIN/(PO_4-P)) was 5.4. The existing water quality data for the Wild Rice River indicated no major water quality problems for proposed Twin Valley Lake.

National Eutrophication Survey Data

29. In 1972, the EPA initiated the National Eutrophication Survey (NES) to investigate the threat of accelerated eutrophication to freshwater lakes and reservoirs. From 1972 to 1976, 815 lakes and reservoirs were surveyed to determine physical and chemical characteristics; trophic state; nutrient sources, loads, and controllability; and limiting nutrients. In Minnesota, 78 lakes were surveyed during 1972 and in North Dakota, 14 lakes were surveyed during 1974. Although each lake was sampled only three times during the year, the number of lakes surveyed make a reasonable data base to compare and evaluate the water quality and eutrophication potential of proposed Twin Valley Lake.

30. Selected physical, chemical, and biological parameters taken from the NES data are summarized in Tables 2 and 3 for 27 lakes and reservoirs located within approximately 200 km of the proposed project. Most of the parameters are self-explanatory. Ranges are given for specific conductance, Secchi disk depth, alkalinity, pH, N/P ratio, and chlorophyll *a*. Specific definitions for symbols and nonstandard terminology are given below:

a. Lake type - N corresponds to a lake of natural origin and I corresponds to a man-made lake (impoundment) created by impounding a stream or river.

b. Stratification - the strength of stratification is related to the maximum temperature difference between the surface and bottom waters.

c. Minimum DO - the minimum DO measured anywhere in the lake.

d. N/P ratio - the ratio of the mean in-lake nitrogen concentration (TSIN) to the mean in-lake dissolved phosphorus or orthophosphate concentration.

e. Limiting nutrient - determined by the Algal Assay Procedure Test (EPA 1971). The symbol U indicates unknown.

f. Chlorophyll *a* - the J indicates that the values may be in error by ± 20 percent due to instrumentation problems in 1972.

g. Trophic state - determined by comparing the measured phosphorus loading rate with those proposed by Vollenweider (1975). The symbols E, M, and O correspond to eutrophic, mesotrophic, and oligotrophic, respectively.

Specific information on NES sampling methods, analytical procedures, etc., can be found in EPA (1974, 1975).

31. The lakes listed in Table 2 were restricted to those located within 200 km of the proposed project to minimize geographical, hydrological, and meteorological differences. The majority of the lakes were east of the proposed project. Morphometrically, the lakes varied in surface area from 28 to 45,300 ha and in mean depth from 1.2 to 13.1 m. Of the 27 lakes listed, 15 had surface areas greater than proposed Twin Valley Lake (218 ha) and 16 had greater mean depths (i.e., Twin Valley Lake = 4.2 m).

32. Proposed Twin Valley Lake differs from the lakes listed in Table 2 in two respects. First, all of the Minnesota lakes are natural lakes. In many instances, natural lakes and reservoirs differ significantly (Baxter 1977). Second, the mean hydraulic residence time for proposed Twin Valley Lake is 0.06 years. This time is less than the 0.1 to 12.7 years listed in Table 2. The six North Dakota reservoirs had residence times greater than 0.65 years.

33. A relative measure of the light penetration in a lake is the Secchi disk depth. The Secchi disk depths in Table 2 varied from 0.3 to 4.3 m. The largest variation in any one lake was 0.1 to 2.9 m. When compared with natural lakes, the impoundments were characterized by smaller minimum Secchi disk depths. Seasonal variations within impoundments and lakes were similar. Proposed Twin Valley Lake should follow this pattern.

34. The strength of thermal stratification was determined by the maximum temperature difference (ΔT) between the surface and bottom waters. All of the lakes, except one, with mean depths greater than 5 m were strongly stratified (i.e., $\Delta T > 5^{\circ}C$). Of 10 lakes with mean depths less than 5 m, 6 did not stratify or were only weakly stratified (i.e., $\Delta T < 2^{\circ}C$). Three of the four remaining lakes that did stratify strongly had surface areas less than 75 ha and possibly were sheltered from the wind. Since proposed Twin Valley Lake would have a mean depth of 4.2 m, a surface area of 218 ha, and a theoretical hydraulic residence time of 0.06 years, it probably would not stratify strongly.

35. Closely related to thermal stratification were the existence and extent of anoxia. Anoxic conditions were assumed to occur when the minimum DO dropped below 2 mg/ℓ. Except for one lake, the strongly stratified lakes (i.e., $\Delta T > 5^{\circ}C$) were anoxic. All of the weakly stratified lakes had DO in the bottom waters. Based on this information, proposed Twin Valley would probably not have DO problems.

36. Alkalinity and specific conductance increased in a westerly direction. Alkalinity varied from 93 to 730 mg/ℓ. Nineteen of the lakes had alkalinities greater than 150 mg/ℓ and were classified as hardwater lakes. The pH varied from 6.5 to 9.4, but most lakes were in the range 7.2 to 8.7. Specific conductance varied from 150 to 2900 μmhos/cm. The conductivities of 22 of the lakes were greater than 250 μmhos/cm. Eight of these had conductivities greater than 500 μhmos/cm. If TDS is assumed to equal 0.6 times the specific conductance, then a typical TDS concentration in the lakes would be greater than 150 mg/ℓ. Because proposed Twin Valley is located near the edge of the moraine (Figure 2) and its watershed extends easterly, the chemical composition of the proposed lake would probably be similar to the natural lakes found in and near its watershed. The project would probably not be as alkaline as the North Dakota reservoirs.

37. Limitations of algal growth in the NES lakes were determined by algal bioassays and N/P ratios. The bioassays indicated nitrogen to be limiting in 12 lakes, phosphorus to be limiting in 4 lakes, and both nitrogen and phosphorus to be limiting in 2 lakes. Results for eight

lakes were not available because of sample preservation problems in 1972. The N/P ratios also indicated nitrogen limitation in most lakes. If an optimum N/P ratio of 15/1 was used, then all of the lakes, except one, had an N/P ratio at some time less than 15 and were nitrogen limited. The one remaining lake had an N/P ratio of 17/1. In 23 lakes the N/P ratios were available for two or more sampling dates. In 12 of these lakes, the maximum ratio was below 15/1. Generally, the larger ratios occurred in spring or early summer. It is therefore possible that proposed Twin Valley Lake would also be nitrogen limited at some time.

38. Total and accumulated annual phosphorus and nitrogen loads were normalized to represent average conditions (EPA 1975). These were used to determine the trophic status of the lakes. Based on Vollenweider (1975), 23 of the lakes were eutrophic; 1 was meso-eutrophic; and 3 were mesotrophic. Since Vollenweider assumed phosphorus limitation and since a majority of the lakes were nitrogen limited, the trophic status in Table 2 should be viewed with caution. Considering only those lakes where phosphorus or phosphorus and nitrogen were limiting, five of six were eutrophic. Proposed Twin Valley Lake would probably also be eutrophic.

39. Chlorophyll a concentrations varied from 0.9 to 130 μg/ℓ. A majority of the lakes had concentrations greater than 20 μg/ℓ. Gakstatter et al. (1975) found that lakes with chlorophyll a concentrations over 10 μg/ℓ were eutrophic. Based on this criteria, 23 of the lakes tested in Table 2 were eutrophic. Chlorophyll a concentrations on the order of 20 to 50 μg/ℓ would probably occur in proposed Twin Valley Lake.

40. Overall, the phytoplankton population consisted of 74 percent Cyanophyta (blue-green algae), 20 percent Chrysophyta (yellow-brown algae, diatoms), 5 percent Chlorophyta (green algae), and 1 percent other, including Pyrrophyta. Seasonally, Chrysophyta and Chlorophyta dominated in April; Cyanophyta dominated in July and September; and Cyanophyta and Chrysophyta dominated in October. Similar trends are expected in proposed Twin Valley Lake.

41. In summary, the NES data set provided an extensive survey of morphometrically different lakes in the vicinity of proposed Twin Valley Lake. Based on the survey, the proposed lake would probably be weakly stratified with DO distributed throughout the water column. Nitrogen would probably be limiting at least some of the time. Phytoplankton blooms on the order of 20 to 50 μg/ℓ chlorophyll a are probable with Cyanophyta being the dominate group. This would be considered eutrophic.

Other Surrounding Impoundments

42. Several other lakes or impoundments in the area of proposed Twin Valley Lake that were not included in the NES data but have been studied include North and South Twin Lakes (MPCA 1976), Otter Tail, Blanche, Walker, and Deer Lakes (MPCA 1969), Dayton Hollow Reservoir (National Biocentric, Inc. 1976), and Lake Orwell (Anderson 1975). Unfortunately, outside of the NES data and selected studies, there is little scientific information on most Minnesota lakes (Limnological Research Center 1976).

43. The lakes and impoundments listed above are similar in morphometry and water quality to those sampled in the NES program. The lakes vary in surface area from 110 to 5600 ha and in maximum depth from 4.5 to 38 m. Half of the above lakes had surface areas comparable to proposed Twin Valley Lake and half were greater in surface area.

44. None of the lakes experienced anoxic conditions in the hypolimnion, although the dissolved oxygen dropped below 2 mg/ℓ on several occasions in Dayton Hollow Reservoir (National Biocentric, Inc. 1976). The other lakes were only sampled on 1 or 2 days so any anoxic conditions that might have occurred could have been missed. The Secchi disc transparencies ranged from 0.5 to 2.1 m. This is comparable with the bottom 14 percent of Secchi disc transparencies measured on other Minnesota lakes (Limnological Research Center 1976).

45. The total phosphorus values within the pool varied from 0.03 to 0.50 mg/ℓ. The higher total phosphorus concentrations were found in

Dayton Hollow Reservoir and were associated with the waste treatment facility above the project at Fergus Falls (National Biocentric, Inc. 1976). Nitrate and ammonium values ranged from 0.0 to 0.4 and from 0 to 0.4 mg/ℓ, respectively. The higher values were again associated with Dayton Hollow Reservoir. The phytoplankton composition is also similar to that found in the NES lakes. The Cyanophyta (blue-greens) dominated the phytoplankton assemblage, comprising 79 percent of the annual standing crop (numbers/ml). The Chlorophyta (greens) and Chrysophyta (yellow-browns, diatoms) comprised 8 and 12 percent of the annual standing crop, respectively. The Pyrrophyta (dinoflagellates) and others again comprised approximately 1 percent. The lakes can be considered moderately alkaline with alkalinity values ranging from 148 to 220 mg/ℓ.

46. The nutrient loadings and productivity estimates in Dayton Hollow would suggest the development of anoxic conditions. However, two factors ameliorate the situation (National Biocentric, Inc. 1976). First, the reservoir only stratified intermittently and then for short periods. Second, the annual hydraulic residence time was 4.4 days, indicating rapid water exchange.

47. Generally, the lakes and reservoirs in the vicinity of proposed Twin Valley Lake can be considered eutrophic. Proposed Twin Valley Lake will probably be of similar trophic status.

PART IV: ALGAL BIOASSAYS

48. Algal bioassays were conducted on water samples collected at the proposed Twin Valley project site to determine the nutrient(s) potentially limiting phytoplankton growth and the availability of the nutrients for plankton uptake. The analyses were performed by the Utah Water Research Laboratory (UWRL), Utah State University, Logan, Utah (Appendix A). This information was required to determine if update data required modification and to select appropriate coefficients for the mathematical ecological simulations.

49. An assumption of the mathematical ecological model used in this study was that all nutrient inputs were in a 100 percent available form for phytoplankton uptake. If only a fraction of a nutrient, e.g., phosphorus, was available for growth, then the input data should be modified to reflect this availability. In addition, model coefficients, such as half-saturation coefficients, must be selected to reflect nutrient concentrations and the limiting nutrient. The various nutrient-loading models used to predict trophic status also require a priori knowledge of the limiting nutrient.

50. Water samples were collected at the proposed project and at Dayton Hollow Reservoir by U. S. Army Engineer Waterways Experiment Station (WES) and U. S. Army Engineer District, St. Paul (NCS) personnel on 10-11 April and by NCS personnel on 11 July 1978. The specific sampling locations were:

- a. Wild Rice River at Twin Valley Gage (Sample No. 1).
- b. Wild Rice River inflow to Lower Rice Lake (Sample No. 2).
- c. Wild Rice River near Faith, Minnesota (upper end of the proposed conservation pool) (Sample No. 3).
- d. Dayton Hollow Reservoir depth integrated sample at midpool (Sample No. 4).
- e. Ottertail River inflow to Dayton Hollow Reservoir (Sample No. 5).

51. The sampling locations on the Wild Rice River are shown in Figure 2. Dayton Hollow Reservoir is located near Fergus Falls, Minnesota (Figure 1). Dayton Hollow Reservoir was selected as a

sampling site to compare in-lake nutrient concentrations and availability with riverine conditions. Since Dayton Hollow Reservoir is morphometrically similar to proposed Twin Valley Lake and since both are located on the edge of the moraine, Dayton Hollow Reservoir should provide a good analogy to proposed Twin Valley Lake.

52. Water samples were taken twice during 1978 to determine seasonal variations. The 10-11 April date was selected to represent snowmelt runoff while 11 July was selected to represent summer base flow. Preliminary USGS flow data indicated that the peak discharge of 182 m^3/sec occurred at Twin Valley on 7 April. This event was the largest on record. The flow on the day of sampling (10 April) was 77.8 m^3/sec. The spring sample, therefore, characterized water on the trailing side of the spring runoff hydrograph. The quality of this water may not be the same as that found at the peak or on the rising side of the hydrograph because many water quality parameters do not load proportionally with flow. Dayton Hollow Reservoir was partially ice covered on the day of sampling (11 April).

53. The flow in the Wild Rice River at Twin Valley during the second sampling trip (11 July) was approximately 2.3 m^3/sec. The mean monthly flows for July and August are 5.6 and 2.4 m^3/sec, respectively. The second sample, therefore, characterized the river at late summer base flow. At Dayton Hollow Reservoir, the flow in the Ottertail River was high due to a rainstorm during the preceding week.

54. All samples were packed in ice and shipped by air freight to UWRL. The samples from the first sampling trip were delayed in reaching UWRL; however, since these samples were still partially iced upon arrival, the bioassay analyses were still conducted. Upon arrival at the UWRL the water samples were filter-sterilized to remove indigenous algae and chemically analyzed. The algal bioassays were performed according to EPA (1971) procedures. The results are briefly described in the following paragraphs and presented in detail in Appendix A.

55. The water samples taken in April from the Wild Rice River at Twin Valley and Faith were chemically similar. Orthophosphate concentrations (0.055 and 0.051 mg/ℓ) were within the range of concentrations

reported in Table 1 but greater than the mean concentration. TSIN concentrations (1.47 and 1.48 mg/ℓ) were two times larger than the maximum TSIN concentration reported in Table 1. These two samples were predicted to be phosphorus limited based on both N/P ratios and bioassay results.

56. The water sample taken in April from the Wild Rice River inflow to Lower Rice Lake was totally different. Concentrations of orthophosphate (0.025 mg/ℓ) were much lower. Because this sampling site was located in the upper watershed, the low concentrations probably reflect the nutrient-trapping characteristics of the many lakes and swamps in the area. The N/P ratio of 8.6 indicated possible nitrogen limitation. The bioassay indicated the sample was only partially limited by nitrogen.

57. During April, concentrations of PO_4-P and TSIN in Dayton Hollow Reservoir and in the Ottertail River inflow to Dayton Hollow Reservoir were similar but less than those measured in the Wild Rice River. The N/P ratios for the reservoir and river were 22 and 19, respectively, indicating phosphorus limitation. Both nitrogen and phosphorus were found to be limiting in the bioassay.

58. The water samples taken in the Wild Rice River in July were less fertile than those taken in April. At Twin Valley, PO_4-P was 0.016 mg/ℓ and appeared to be less than adequate for growth. TSIN concentrations ranged from 0.078 mg/ℓ at the Lower Rice Lake inflow to 0.155 mg/ℓ at Faith. These concentrations were one order of magnitude less than those measured in April and resulted in N/P ratios less than 5 (i.e., nitrogen limitation). Bioassays confirmed that the sample taken from the inflow to Lower Rice River Lake was nitrogen limited. The bioassays were inconclusive for the samples collected at Twin Valley and Faith since both nitrogen and phosphorus spikes were required for growth.

59. The July nutrient concentrations in Dayton Hollow Reservoir and its inflow were higher than those measured in the Wild Rice River. Rain over the Ottertail River basin during the preceding week probably accounted for the higher concentrations. The higher concentrations in Dayton Hollow Reservoir compared to its inflow probably resulted from the peak stormflow loadings to the reservoir having already passed by

the sampling site on the river. Both of these samples were predicted to be nitrogen limited based on both N/P ratios and bioassay results.

60. In summary, the chemical analyses and bioassays from Dayton Hollow Reservoir indicated that conditions in the reservoir and river were similar. Therefore, conditions in proposed Twin Valley Lake may be similar to those found in the Wild Rice River. The bioassays from the Wild Rice River indicated that either phosphorus or nitrogen could be limiting and, at times, another constituent such as carbon may limit growth. The probability is higher, however, that phosphorus may be limiting in proposed Twin Valley Lake since some phosphorus may co-precipitate out of the water column with $CaCO_3$ and because many blue-green algae have the ability to fix nitrogen. The bioassays also indicated that the nutrients were in a completely available form and that the waters of the Wild Rice River were relatively infertile.

61. It is important to realize that the algal bioassays reflect only the conditions occurring in the lacustrine or riverine system at that instant. The limiting factor may change through time to some other constituent such as carbon or light. Light is limiting in many reservoirs, for example, due to the high suspended sediment loads and resuspension of deposited sediments due to increased wind velocities. In addition, the algal bioassays reflect the requirements of a single phytoplankton species. Since phytoplankton species do not exist as isolated entities but as competitive, interacting members of an assemblage in a natural system, care must be exercised in extrapolating the results to the prototype system. The purpose of the bioassays was not to provide an absolute, time-invariate answer concerning limiting factors, but rather to provide insight into those factors that may limit phytoplankton growth during different seasons and the relative magnitude of this limitation.

PART V: MATHEMATICAL SIMULATIONS

62. A research version of the Water Quality for River-Reservoir Systems (WQRRS) reservoir model was used in this study. The model was originally modified for the Hydrologic Engineering Center, Davis, California, by Water Resources Engineers (WRE), Walnut Creek, California. This model has been applied to a variety of Corps of Engineers (CE) project studies in the past (Thornton et al. 1976; Thornton et al. 1977a; Hall et al. 1977; Ford et al. 1977; Ford et al. 1978; and others). Since the 1977 documentation of the WQRRS model (Hydrologic Engineering Center 1977), several significant modifications have been incorporated into the model by the Environmental Laboratory at WES. A partial listing of these is presented in Table 4. These modifications were made to produce more realistic simulations of reservoir ecosystems. Before results of the model simulations made for Twin Valley Lake are discussed, it is important to examine and understand the major assumptions and limitations of the WQRRS model since these play an important role in the interpretation process.

Major Assumptions and Limitations of the WQRRS Model

63. The major assumptions and limitations of the WQRRS model are as follows:

a. A reservoir can be represented by a vertical series of one-dimensional horizontal slices. This implies that only the vertical dimension is retained during computation. Each horizontal layer is assumed to be completely homogeneous. Therefore, all isotherms and all isopleths of water quality constituents such as DO, nutrients, and algae or other biota are parallel to the water surface both laterally and longitudinally. In addition, all inflow and outflow quantities and concentrations are instantaneously dispersed and homogeneously mixed throughout each horizontal layer from the headwaters of the impoundment to the dam. It is not possible, therefore, to look at longitudinal variations in constituents such as coliforms or phytoplankton.

b. In general, the vertical dimensions are specified for the deepest portion of the reservoir, and the model results will be most representative of conditions in this area. Since this area is usually near the structure, it is difficult to draw conclusions on water quality conditions expected to occur in coves, embayments, or the headwater areas. Similarly, the specification of initial conditions is for the deepest area of the reservoir.

c. Vertical placement of inflowing water within the impoundment is determined by temperature only. The density of an inflow is determined from its temperature, and it is placed into the horizontal zone of comparable density. Contributions to the density of the inflow by suspended and dissolved solids are not currently included in the model. These contributions may not be significant in this study, however. In the headwaters, both deposition and inflow mixing occur. The effect of each of these processes is to decrease the contribution of suspended solids to the density of the inflow. In addition, calculations show that the density contribution of suspended solids concentrations up to 1000 mg/ℓ is less than the density change resulting from a 1°C decrease in temperature at 25°C.

d. Internal dispersion of thermal energy and mass is accomplished by an effective diffusion mechanism that combines the effects of molecular diffusion, turbulent stirring and mixing, and thermal convection. The transport is therefore assumed to be proportional to an effective diffusion coefficient and a concentration gradient. It is important to note that although the diffusion gradient among layers is based on the concentration differences of the individual constituents such as DO or nitrate, the effective diffusion coefficient is always based on temperature. In many instances, mass diffusion coefficients may not be equivalent to thermal diffusion coefficients.

e. The dynamics of each chemical and biological component can be described by Conservation of Mass and the Kinetic Principle. The mass of elements such as carbon, oxygen, nitrogen, and phosphorus is accounted for by considering the inflows, outflows, and internal changes in the form of the elements. The Kinetic Principle implies that these internal changes occur through rate processes such as uptake, decay, respiration, etc.

f. The chemical and biological rate processes occur in an aerobic environment. While the WQRRS model does have some simple default algorithms that are called when DO is zero, the model predictions are not realistic under anoxic or zero DO conditions. The model algorithms were developed to simulate the rate processes and reactions occurring under

aerobic conditions. This limitation results in an inability to simulate the buildup of a DO debt under anaerobic conditions. Consequently, the model may predict the timing and rate of DO increases following the end of anaerobic conditions to be greater than might actually occur. Also, changes in the solubility and formation of various chemical species and interactions between sediment and water under anaerobic conditions cannot be simulated.

g. The model does not contain an ice cover algorithm. Since conditions under an ice cover cannot be simulated, model predictions are limited to ice-free periods.

Selection of Study Years

64. Three years representing dry, wet, and average conditions were selected to simulate the effects of hydrometeorological conditions on project water quality. Based on 55 years of record (1910-1917, 1931-1977) at the Twin Valley gage, the dry, wet, and average study years were selected to be 1976, 1975, and 1971, respectively.

65. The selection criteria included:

a. Years with water quality data (1975-77) were considered first.

b. If it was impossible to select a year where water quality data existed, then a year was selected as close as possible to the years where data existed.

c. Consideration was given to the distribution of mean monthly flows for the period of open water (March through November) as well as to the ranking of the mean annual flows.

d. The wet year was selected to represent conditions one standard deviation above mean conditions.

e. The dry year was selected to represent conditions one standard deviation below mean conditions.

66. The mean annual flow at Twin Valley for the period of record was 5 m^3/sec with a standard deviation of 2.8 m^3/sec. Therefore, the wet year should have a mean annual flow of 7.8 m^3/sec and the dry year should be 2.2 m^3/sec. The mean annual flows for the study years, 1976, 1975, and 1971, were 3.1, 10.4, and 5.4 m^3/sec, respectively. The second wettest year on record was 1975, while 1976 ranked in the lower

half of the dry years. With the exception of 1958, the drier years with ranking in the 25th percentile or below occurred prior to 1940. It would be impossible to include any of these years in this study, since current land-use practices in the watershed affecting water quality do not reflect pre-1940 conditions.

67. The mean monthly flows for the study years were compared with the mean monthly flows for the period of record in Figure 23. The mean monthly flows for 1971 were near the long-term mean for March, April, and September (Figure 23a). They were below average for the period May-July and above average in October. The April through July mean monthly flows for 1975 exceeded the long-term mean plus one standard deviation. Only in the months of March, September, and October did the mean monthly flows fall below the long-term mean (Figure 23b). In 1976, the mean monthly flows could be considered representative of dry conditions with the exception of the period of spring runoff (March through April) where the flows were near or above normal (Figure 23c).

68. Considering criteria a and b above and the fact that mean conditions are seldom realized when dealing with natural phenomena, 1971, 1975, and 1976 were considered representative of average, wet, and dry conditions, respectively.

Data Requirements

69. Data requirements for the ecological model can be categorized into initial conditions, coefficients, and updates.

Initial conditions

70. Initial conditions refer to those that exist in a lake at the time the simulations are started. Ideally, it would be desirable to simulate an entire year. This was impossible, however, because the model could not simulate the effects of ice and snow cover (paragraph 63). The simulations, therefore, were started at the time of ice breakup. At this time, specification of initial conditions was simplified because the lake could be assumed to be isotropic (well mixed) due to spring turnover.

71. Little information on the time of ice breakup was found for lakes in the vicinity of proposed Twin Valley Lake. Dates of ice out were available from Ford (1976) for natural lakes located in the Minneapolis-St. Paul metropolitan area. The dates for the study years 1971, 1975, and 1976, were 15 April, 26 April, and 28 March, respectively. Since Minneapolis-St. Paul (latitude 45^o) is south of the proposed project (latitude 47.3^o), ice out would probably be expected to occur later at Twin Valley. This may not necessarily be true because, in general, the ice usually goes out sooner on reservoirs than natural lakes due to the large inflows during the spring thaw.

72. The spring flood peaked at Twin Valley on 10 April 1971, 19 April 1975, and 28 March 1976. These dates are within 1 week of the dates of ice breakup of natural lakes in Minneapolis-St. Paul. The temperature of the Wild Rice River on 19 April 1975 was 1^oC. Considering this temperature, the heat of fusion of water, the large flows, and the general condition of the ice during this time of year, it is probable that the ice would break up with the spring flood or shortly thereafter.

73. It was therefore assumed that the period of ice breakup coincided with the peak of the spring flood. Simulations were started on 10 April 1971 (day 100), 19 April 1975 (day 109), and 28 March 1976 (day 87). This assumption has the additional advantage of including at least part of the spring runoff in the simulations. Since a major portion of the loadings to a reservoir may occur during this event, it is important to include as much of the event as possible.

74. During the peak of the spring flood, mean residence times in the proposed impoundment would be 3 to 4 days. Inflow densimetric Froude numbers would be greater than one indicating plug flow. Changes in the flow regime from river to reservoir would therefore be negligible and conditions within the proposed reservoir would be similar to conditions in the river. Initial conditions for the proposed reservoir were therefore assumed to be similar to those in the Wild Rice River at the start of the simulations. This assumption was substantiated with data taken from the Ottertail River and Dayton Hollow Reservoir

(paragraph 57) where nutrient concentrations in the lake and river were similar.

75. The initial conditions for the 3 study years are given in Appendix D. Initial temperatures were assumed to equal the average equilibrium temperature for the preceding 10-day period. The DO concentrations were assumed to be saturated at these temperatures. Initial standing stock estimates for the three fish compartments and for the benthos and zooplankton compartments were obtained from Peterson (1976) and Peterka and Knutson (1970). Algae concentrations were determined from data on the Wild Rice River (USGS 1976, 1977) and data on surrounding lakes (paragraph 42). Organic sediment was calculated from the soil samples used in Appendix B. Initial conditions for all other constituents were assumed equal to those measured in the Wild Rice River at the time the simulations were started.

Coefficients

76. Coefficients required to simulate proposed Twin Valley Lake ranged from those that are physically definable and well documented in the literature to those that are difficult to measure in either the laboratory or the field and must be quantified, based primarily on the experience of the investigator and calibration runs. The coefficients used in the base simulations are listed in Appendix D.

77. The coefficients that determine the hydrothermal regime were determined by calibrating the model on Lakes Calhoun and Turtle. Although these two lakes are located a substantial distance from the project, they do have surface areas similar to proposed Twin Valley Lake and the mixing and heat budget coefficients should also be similar. Differences in macroclimate between the two locations are taken into consideration by the meteorological updates. The calibration is discussed in detail in a later section on thermal simulations.

78. Since the ecological model has the capability to simulate the phytoplankton dynamics with only two compartments, the species of phytoplankton listed in Table 5 were aggregated into two groups. Lewis (1977) found strong correlations in net growth rate among abundant species at intradivisional levels. Using this criteria and the fact that the blue-green algae contributed 75 percent of the phytoplankton

standing crop (numbers/ml), one compartment represented the cyanophytes or blue-green algae (ALGAE 2). The chlorophytes and chrysophytes or green algae and diatoms were aggregated to represent the ALGAE 1 compartment. The diatoms and greens represented 20 and 5 percent of the phytoplankton standing crop, respectively. The green algae and diatoms were aggregated since their seasonal phasing was similar and many of their environmental requirements were comparable. The pyrrophytes were not included in the aggregations since (a) their standing crop represented only 1 percent of the total and (b) little information existed to obtain the various required rate coefficients and constants. The list of species considered during compartmentalization and the simulations is shown in Table 5. The species represent 99 percent of the standing crop within their respective divisions.

79. Growth, respiration, and settling rates and nitrogen, phosphorus, carbon, and light half-saturation coefficients for the compartmental assemblages based on the species listed in Table 5 were compiled from the literature (Appendix E). These rates and coefficients were weighted by dominance (percent of compartment abundance) and the number of occurrences in the surrounding impoundments to obtain composite rates for each aggregation.

80. Zooplankton and benthos were each represented by only one compartment in the model. The composition of the zooplankton and benthos assemblages in the surrounding impoundments was obtained from Peterka and Knutson (1970), MPCA (1971), MPCA (1969), and Anderson (1975). Coefficients were selected from the literature (Appendix E) to represent these compositions.

81. The fishery compartments in the model are capable of simulating planktivorous, benthic-feeding, and piscivorous fishes. Based on information from Peterson (1976) on fish production and composition in the Wild Rice River watershed, the fisheries were apportioned among the three compartments. The planktivorous compartment was represented by percentages of shad, sucker, crappie, and buffalo standing crop; benthic feeders by carp, suckers, sunfish, catfish, and rock bass; and piscivores by crappie, northern pike, catfish, yellow perch, and sauger. Coefficients for these respective compartments were obtained from Leidy and

Jenkins (1977) and weighted to reflect the fisheries composition in each compartment.

82. Other coefficients such as decay rates for BOD and NH_4-N, etc., were taken from Sawyer and McCarty (1967), Kittrell and Furfari (1963), previous simulations, and other literature cited in Appendix E.

Updates

83. Updates refer to model inputs that vary with time. These include meteorological data, flow data, and inflowing constituent concentrations.

84. The meteorological data were obtained from the Class A weather station located at Fargo, North Dakota, approximately 65 km southwest of the proposed project. The 3-hr data for cloud cover, air temperature, dew point temperature, barometric pressure, and wind speed were averaged over 24 hr to provide daily updates.

85. Daily inflows to the reservoir were obtained from the USGS gage at Twin Valley. The 1971, 1975, and 1976 discharges are shown in Figure 24. The inflows were routed through the reservoir by NCS assuming a maximum release of 70 m^3/sec. Project operation of the selective withdrawal structure was selected to meet a natural temperature objective downstream. This operation is discussed in detail in a later section on thermal simulations.

86. Daily water temperatures were available at Twin Valley for days 100-119 and 128-144 in 1975, 140-305 in 1976, and 60-304 in 1977. Water temperatures were generated from daily air temperatures for 1971 and for 1975 and 1976 to fill in missing data. The generation technique assumed that both water and air temperature could be divided into a harmonic component and a residual. That is,

$$T_w = T_{wh} + T_{wr} \qquad (1)$$

where T_w = water temperature, °C

T_{wh} = harmonic water temperature, °C

T_{wr} = water temperature residual, °C

and

$$T_a = T_{ah} + T_{ar} \quad (2)$$

where T_a = air temperature, ^oC

T_{ah} = harmonic air temperature, ^oC

T_{ar} = air temperature residual, ^oC.

The water temperature residual was then assumed to be related to the air temperature residual by the following expression

$$T_{wr}(t) = aT_{ar}(t) + bT_{ar}(t-1) + cT_{ar}(t-2) + dFlow \quad (3)$$

where

t = time, days

a, b, c, d = coefficients

Flow = river discharge, m^3/sec

Using data from 1975 and 1976 and a least squares fit of Fourier coefficients, the first order harmonic for water temperature was determined to be

$$T_{wh}(t) = 5.82 - 19.57 \cos\left(\frac{2\pi}{365} t - 0.33\right) \quad R^2 = 0.91 \quad (4)$$

and the first order harmonic for air temperature was determined to be

$$T_{ah}(t) = 5.32 - 18.22 \cos\left(\frac{2\pi}{365} t - 0.30\right) \quad R^2 = 0.80. \quad (5)$$

A multiple linear regression model was used to obtain the following equation for water temperature:

$$T_w(t) = 1.66 + 0.92T_{wh}(t) - 0.028T_{ar}(t) + 0.238T_{ar}(t-1) \quad (6)$$

$$+ 0.228T_{ar}(t-2) - 0.057 \text{ Flow}$$

The coefficient of determination (R^2) was 0.97 and the standard deviation was 1.4^oC.

87. Water quality updates for the 3 study years are summarized in Appendix D. Updates for alkalinity, BOD5, NH_4-N, NO_2-N, NO_3-N, fecal coliforms, PO_4-P, TDS, and pH were obtained from linear interpolation between data. If no data existed, as was the case for 1971 and some

parameters in 1975, updates were generated from regression equations based on inflow or assumed equal to the mean value given in Table 1. In all 3 years, DO was assumed to be 94 percent saturated (paragraph 16).

88. Based on a comparison of data collected by the USGS (1976, 1977) in the Wild Rice River and data collected on surrounding lakes, it was assumed that no inputs of ALGAE 2 and zooplankton from the river to the reservoir would remain viable. The stream inputs for these variables were considered to be included in the detritus updates. A fraction of the ALGAE 1 compartment was assumed to remain viable in the reservoir and was included as a constant update. The fraction that was assumed not to be viable was included in the detritus updates.

Thermal Simulations

89. Thermal simulations were used to calibrate the mixing and heat transfer coefficients required for the ecological model simulations; to evaluate the selective withdrawal capabilities of the project with respect to a downstream temperature objective; and to determine if proposed Twin Valley Lake would stratify.

Calibration of thermal structure

90. The thermal model was calibrated on two natural lakes (viz, Calhoun and Turtle) in the Minneapolis-St. Paul metropolitan area. Although these lakes were located approximately 360 km southeast of the proposed project, they were similar in surface area (i.e., 180 ha and 170 ha versus 218 ha). Mixing resulting from energy inputs through the air-water interface (i.e., wind and natural convection) should therefore be similar in the three lakes. Natural lakes were used in the calibration to eliminate the mixing resulting from inflow-outflow. This mixing was treated separately by the model and including its effects in the calibration only complicated the process. Two lakes were used to demonstrate the capability of the model to simulate both deep, strongly stratified lakes (Calhoun) and shallow, weakly stratified lakes (Turtle).

91. The temperature data used to calibrate the model were obtained from Ford (1976). Meteorological data came from the Class A station

located at the Minneapolis-St. Paul International Airport. The simulations were started with measured temperature profiles on day 113. The calibration results are shown in Figures 25 and 26. The model was able to simulate the onset and destruction of the strong stratification in Lake Calhoun. Simulated temperatures were within 2°C of measured data and the thermocline depth also agreed. The simulation of Turtle Lake exhibited too much stratification in early summer. Fall overturn was simulated to occur at the correct time. Since proposed Twin Valley Lake would also be shallow, predictions of stratification should be conservative.

Temperature objective

92. As currently planned, the temperature of downstream releases would be controlled through the operation of a multilevel selective withdrawal structure. The extent of control would, however, depend on the in-lake temperature structure. In this study, the project was operated to maintain natural stream temperature.

93. The natural temperature cycle was determined by fitting a first order harmonic to the temperature data measured at Twin Valley from 1975 to 1977. The result was

$$T = 5.82 - 19.57 \cos\left(\frac{2\pi}{365} t - 0.33\right) \tag{7}$$

where T = objective temperature, °C

t = time (Julian day).

The coefficient of determination was 0.87. This idealized temperature objective is compared with measured data in Figure 27 to illustrate the degree of scatter that can be expected.

Selective withdrawal capabilities

94. The selective withdrawal capabilities of the Twin Valley project were evaluated using the thermal compartment of the ecological model and selective withdrawal routines developed by the WES Hydraulics Laboratory. Simulations were made to determine the feasibility of meeting the downstream temperature objective.

95. The selective withdrawal structure described by R. W. Beck and

Associates (1978) was used in the initial simulations. The proposed structure consisted of two water quality gates and two flood control gates. The specifics of the withdrawal ports are given in the following tabulation:

Port No.	Center-Line Elevation m msl	Size, m	Minimum Flow Capacity m^3/sec	Maximum Flow Capacity m^3/sec
1	322.5	1.37 × 1.83	0.14	7.64
2	321.2	1.37 × 1.83	0.14	7.64
2 flood-gates	317.0	1.37 × 3.05	1.42	70.08

The structure is shown in Figure 28 to illustrate the relative location of the ports.

96. Simulations were made for the 3 study years to determine if the natural temperature objective could be met with this structure. Considering the natural scatter illustrated in Figure 27, the downstream objective was met in all 3 years (Figure 29). The port operations required to meet the objective are shown in Figure 30. Except for periods of flood control operations, the releases were mainly through the upper port.

97. To determine the sensitivity of the project to meeting the downstream temperature objective, surface and bottom withdrawal were also simulated. It was assumed that during surface withdrawal all of the flow, even the flood releases, could be passed through the top port. It was also assumed that during bottom withdrawal, the flow could be controlled down to 0.14 m^3/sec. From a practical standpoint, both of these assumptions are unrealistic considering the structures described in paragraph 95, but they do represent two extremes. Figures 31 and 32 show that the temperature objective was met using surface and bottom withdrawal. Therefore, selective withdrawal is not required to meet the downstream temperature objective.

Thermal predictions

98. Using the coefficients obtained from the calibration simulations and assuming selective withdrawal capabilities, the thermal

structure of proposed Twin Valley Lake was simulated for the 3 study years. The strongest stratification was exhibited in 1971, lasting from early May through the middle of July (Figure 33). During this period of stratification, mean monthly flows in the Wild Rice River and wind speeds at Fargo were abnormally low while mean monthly air temperatures were near or above normal. The more dynamic years, 1975 and 1976, exhibited intermittent periods of weak stratification (Figures 34 and 35). Typically, these remained from 5 to 20 days. Abnormally high flows in 1975 and abnormally high winds in 1976 were responsible for keeping the lake well mixed. In all three years the lake remained well mixed after it achieved maximum thermal energy content and started to cool. This usually occurred in late July or early August.

99. Zones of inflow and outflow for the study years are shown in Figures 36-38. In 1971, the inflow was distributed throughout the water column except during periods of stratification where it entered the metalimnion. The outflow was restricted to surface waters during periods of stratification. In 1975, except for isolated pockets, the inflows were distributed throughout and the outflows were pulled from the entire water column (Figure 37). In 1976, the inflows and outflows were generally restricted to the surface waters.

Water Quality Simulations

100. Deterministic and stochastic water quality simulations were made of proposed Twin Valley Lake to predict its water quality and trophic status and determine its sensitivity to various coefficients and update parameters. Although 21 water quality variables were simulated, only temperature, DO, algae, and fecal coliforms are of primary interest and will be discussed. The remaining variables will be used to explain variations in these parameters. The simulation results will be discussed according to deterministic simulations, Monte Carlo (stochastic) simulations, and management alternatives.

Deterministic simulations

101. Deterministic simulations were made of the 3 study years to

calibrate the model and to check coefficient sensitivity before the Monte Carlo simulations were conducted. After several sensitivity runs, it was determined that the initial coefficients were acceptable and no calibration was necessary. These coefficients are summarized in Appendix D.

102. 1971. Six algae blooms were predicted to occur in 1971 (Figure 39). The first bloom, ALGAE 1, started soon after the simulation was initiated and persisted for approximately 40 days, attaining a maximum concentration of approximately 2 g/m^3. Nitrogen was predicted to be limiting through most of the bloom. A small ALGAE 2 bloom immediately followed the ALGAE 1 bloom. Its magnitude was limited by the first bloom. Three large ALGAE 2 blooms dominated the summer. The blooms persisted for periods of 15 to 25 days and attained concentrations up to 9 g/m^3. Although the first bloom was initially limited by phosphorus, carbon limitation dominated through most of the summer. A small ALGAE 1 bloom (ca 1 g/m^3) was predicted to occur in the fall.

103. DO was principally controlled by algal blooms and thermal stratification. Generally, the DO in the surface waters responded inversely to temperature (Figure 40). DO maxima (ca 130 to 140 percent saturation) occurred during phytoplankton blooms (e.g., days 188 and 224) and minima (ca 55 to 60 percent saturation) occurred after the bloom (e.g., days 209 and 239). Three distinct periods of anoxia (DO < 2 mg/ℓ) were predicted to occur in 1971 (Figure 41). These periods varied in length from 10 to 85 days and included up to 15 percent of the lake volume. After the lake stratified, anaerobic conditions developed in 5 to 10 days. DO was predicted to return to the bottom waters immediately at turnover.

104. 1975. In the wet year, the first bloom of ALGAE 1 was delayed until the river flow dropped below 30 m^3/sec (Figure 42). At the time the bloom started, the theoretical hydraulic residence time of the lake was 7 or 8 days and carbon and light were predicted to be limiting algal growth. The magnitude of the first bloom was 4 g/m^3 and it persisted for approximately 50 days. A second bloom of ALGAE 1 started immediately after the first bloom subsided but it was overtaken by an

ALGAE 2 bloom. ALGAE 2 had the competitive advantage because nitrogen was predicted to be limiting. The magnitude of the ALGAE 2 bloom was 7 g/m^3 and it persisted about 25 days. Carbon and nitrogen were predicted to be limiting during this bloom. The bloom declined rapidly with increased zooplankton grazing. During the second ALGAE 2 bloom carbon was predicted to limit algal growth. Small blooms of ALGAE 1 and ALGAE 2 occurred in the fall.

105. As in 1971, the DO of the surface waters responded to phytoplankton blooms and the bottom 2 or 3 m went anoxic when the lake stratified (Figure 43). Although the lake was predicted to turn over several times during the spring and early summer, anaerobic conditions may persist from day 130 to day 230. A few days of mixing may not be sufficient to overcome the oxygen debt that has built up.

106. <u>1976.</u> Four major algae blooms were predicted to occur in the dry year (Figure 44). The first bloom of ALGAE 1 began on day 101 when the river flow dropped below 20 m^3/sec. Phosphorus was predicted to be limiting during most of this bloom. A small bloom of ALGAE 1 and ALGAE 2 followed. Zooplankton grazing regulated the size of this bloom and contributed to the decline of ALGAE 1. A major bloom of ALGAE 1 was predicted to occur in late July. The bloom persisted for about 2 weeks and had a magnitude of 6.7 g/m^3. This bloom was followed by a large ALGAE 2 bloom which had a magnitude of over 10 g/m^3. Carbon was predicted to be limiting during this last bloom.

107. DO was similar to 1971 and 1975. Two major periods of anoxia were predicted (Figure 45). It is probable, however, that anaerobic conditions may extend from day 115 to day 205. As in the other two years, it took 5 to 10 days after stratification for anoxia to set in.

108. <u>Fecal coliforms.</u> Coliforms were found not to be a problem in any of the study years. All predictions remained below the standard of 200 colonies/100 ml.

<u>Monte Carlo simulations</u>

109. There were insufficient data for an accurate definition of required coefficients and updates for this study. Many of the coefficients also vary with time in a manner that is currently beyond

mathematical description. One way to estimate the uncertainty in the predictions resulting from these and other inadequacies is to vary a coefficient and observe the perturbations. A more rigorous approach is to use Monte Carlo simulations.

110. In Monte Carlo simulations, any number of coefficients and updates can be varied simultaneously within prescribed limits and distributions. Specifically, new values for the coefficients and updates were selected at random from the specified distributions during each computational step. Every simulation was, therefore, different. By superimposing several simulations, it was possible to compute a mean and confidence interval that was time-varying and incorporated the uncertainty and variance in the input data and coefficients.

111. Distributions. Coefficient distributions were determined from data published in the literature (Appendix E). When there were not sufficient data to define a distribution, a uniform distribution was assumed. Limits were established to include all known data. Uniform distributions were assumed for all decay rates, all settling rates, and carbon and light half-saturation coefficients. The distribution of growth rates for the ALGAE 1 species expected to occur in Twin Valley Lake was found to be Gaussian or normal. This was also found to be true for all respiration rates and zooplankton growth and mortality rates. The distribution of growth rates for the ALGAE 2 species, or blue-green algae, resembled a rotated log normal distribution (Figure 46).

112. The distributions for phosphorus and nitrogen half-saturation coefficients were also found to be skewed to the left resembling a rotated log normal distribution (Figure 47). One reason for this skewness could be the ambiguous definition of a eutrophic lake. Since only coefficients from eutrophic lakes were considered and since the half-saturation coefficients increase with nutrient concentration (Hendrey and Welch 1973; Carpenter and Guillard 1971; Toetz et al. 1973), any eutrophic lake characterized by low nutrient concentrations would also be characterized by low half-saturation coefficients. However, a lake considered eutrophic in one part of the country may not be considered eutrophic in another part. The inclusion of coefficients from these

lakes would tend to skew the distribution to the left. Another reason for the sharp decline at large concentrations could be that another nutrient is limiting.

113. The scatter plots of the updates with flow (Figures 7-15) indicated very little, if any, relationship with flow. Updates of ALGAE 1, alkalinity, BOD, ammonium, nitrate, fecal coliforms, detritus, and phosphorus were therefore assumed to be uniformly distributed. Several planktonic diatom genera were indicated by the USGS to be present in the Wild Rice River. Assuming a liberal case, these diatoms were considered to be 100 percent viable when entering the proposed Twin Valley Lake and capable of contributing to the ALGAE 1 assemblage. The genera of blue-green algae found in the Wild Rice River were not indicated in any of the surrounding lakes and were, therefore, included in the detritus updates. Some parameters, such as mass fractions and stoichiometric relationships, were always held constant to prevent mass imbalances from occurring. The coefficients and updates that were varied are summarized in Table 6.

114. The coefficients and update values were selected at random from the distributions during each time step. One loop in the simulations consisted of simulating the interval from ice-out or isothermal conditions in the spring to the ice formation period in the fall. The initial conditions were reset and another loop through the simulation interval was made, again selecting the coefficients at random. A total of 30 loops were made for each simulation condition listed below.

115. Base condition. The base condition has previously been described in the description of the thermal simulations. It consisted of using the selective withdrawal structure proposed for Twin Valley Lake and the operational schedule provided by the St. Paul District.

116. During 1971, the average year, anoxic or low DO conditions developed in the bottom on three occasions (Figure 48). The hypolimnion was anoxic for ca 10 days from 10 May to 20 May, ca 27 days from 3 June to 30 June, and less than or equal to 2 mg/ℓ for ca 14 days from 7 August to 21 August. These periods corresponded to periods of weak thermal stratification.

117. There was a period of increased ALGAE 1 biomass from ca 6 May to 22 May (Figure 49). The maximum mean biomass attained during this increase was 1.8 g/m^3. Using a conversion factor for phytoplankton of 0.23 g/m^3 dry weight to 1 μg/ℓ of chlorophyll _a_ (Spangler 1969), this corresponded to a chlorophyll _a_ value of 7.7 μg/ℓ. A minor increase in biomass occurred around 6 September equivalent to a chlorophyll _a_ concentration of 2.3 μg/ℓ. The ALGAE 2 compartment, representing the blue-green algae, increased around 8 June and remained near or above 1.8 mg/ℓ until approximately 17 October (Figure 50). Blue-green algae have approximately twice the mass per unit of chlorophyll _a_ as other phytoplankton species (Kalff and Knoechel 1978), so this corresponded to a chlorophyll _a_ value of ca 3.8 μg/ℓ. There was a pentamodal increase in the ALGAE 2 assemblage during this period with a mean maximum chlorophyll _a_ concentration of 8.8 μg/ℓ.

118. Anoxic conditions were more prevalent during 1975, the wet year (Figure 51). Dissolved oxygen concentrations were less than 2 mg/ℓ or zero for 14 days from 11 May to 25 May, 10 days from 30 May to 9 June, 11 days from 17 June to 28 June, 14 days from 3 July to 17 July, and 8 days from 28 July to 5 August.

119. The phytoplankton biomass was also greater during 1975 than 1971. The ALGAE 1 assemblage exhibited increases from ca 20 May until 17 June and 17 July until 8 August (Figure 52). The mean maximum biomass during these periods was 4.8 and 2.5 g/m^3, respectively. This was equivalent to chlorophyll _a_ concentrations of 20.9 and 10.7 μg/ℓ, respectively. The increase in both the ALGAE 1 and ALGAE 2 mass occurred later in 1975 than in 1971. The increase in ALGAE 2 biomass was greater and more distinct during 1975 than 1971 (Figure 53). The first increase in biomass occurred from approximately 17 July to 22 August while the second increase occurred from 7 September to 4 October. The mean peak biomass, and equivalent chlorophyll _a_ concentrations, for these two periods were 6.9 and 9.2 g/m^3 and 15 and 20 μg/ℓ, respectively.

120. There were two anoxic periods during the dry year, 1976 (Figure 54). Dissolved oxygen was less than 2 mg/ℓ or zero for 25 days from 21 May to 15 June and 22 days from 6 July to 28 July.

121. The ALGAE 1 assemblage exhibited a trimodal increase in biomass with successively decreased magnitude (Figure 55). The first increase occurred from ca 12 May to 6 June. A second, smaller increase occurred from 25 June to 9 July. There was a very minor increase around 25 July. The mean maximum biomass and chlorophyll a equivalent during the first increase was 3.2 g/m^3 and 13.8 μg/ℓ, respectively. The ALGAE 2 assemblage had three distinct increases (Figure 56). These blooms occurred from ca 25 June to 19 July, 25 July to 26 August, and 1 September to 15 October. The mean peak biomass and equivalent chlorophyll a concentrations associated with these three blooms were 4.6, 6.7, 5.5 g/m^3 and 10.0, 14.6, 12.0 μg/ℓ.

122. The onset of anoxic conditions occurred anywhere from 10 to 21 May during the 3 study years. The first ALGAE 1 bloom occurred sometime during 6 to 20 May while the first ALGAE 2 bloom occurred from 8 June to 17 July.

Management alternatives

123. Several alternative operational approaches were evaluated in this study to assess their impact on water quality, specifically with respect to reducing anoxic conditions and phytoplankton blooms. These operational approaches included bottom and surface withdrawal, lower and higher pool elevations, destratification, increased minimum releases, and decreased maximum releases.

124. Bottom and surface withdrawal. In general, no differences were observed in the water quality of proposed Twin Valley Lake with bottom, surface, or selective withdrawal during any of the study years except for the ALGAE 2 blooms during 1975 (Figures 57-74). With bottom withdrawal, the magnitude, variance, and duration of the blooms during 1975 were increased. The mean peak biomass was 9.35 and 10.9 g/m^3 during the two blooms compared to 6.9 and 9.2 g/m^3 with selective withdrawal, respectively. The peak value in the 95 percent confidence interval was 16.6 g/m^3 with bottom withdrawal versus 13.5 g/m^3 with selective withdrawal. The bloom extended beyond the termination date of 14 November with bottom withdrawal but was completed by 4 October with selective withdrawal. With surface withdrawal, the magnitude, variance,

and duration of the blooms were decreased. The mean peak biomass values for the two blooms were 4.8 and 5.6 g/m^3 with surface withdrawal. The peak biomass value in the 95 percent confidence interval was 6.9 g/m^3 compared with 13.5 g/m^3 with selective withdrawal. Although there was a minor bloom late in the fall, the major blooms had occurred by 28 September with surface withdrawal.

125. With the exception of the ALGAE 2 blooms during 1975, there were no significant differences among the withdrawal schemes for the 3 years. However, 1975 was the second wettest year on record, and, therefore, has a lower probability of recurrence. For the above reasons and the economics associated with a bottom withdrawal versus selective withdrawal structure, most of the additional simulations were compared with the bottom withdrawal results for the 3 study years.

126. Increased and decreased pool elevation. The general trend in changing the pool elevation from 324 m msl during all study years was to improve the bottom DO concentrations slightly by lowering the pool elevation to 322.5 m msl and to decrease the DO slightly by raising the pool elevation to 325.5 m msl (Figures 75-80). The opposite trend occurred with the phytoplankton assemblages. Phytoplankton blooms were enhanced at the lower pool elevation and diminished at the upper pool elevation during all years (Figures 81-92). The distinctions among pool elevations were exaggerated again in the ALGAE 2 blooms during 1975. The phasing of the blooms was similar at all three pool elevations, but at the lower pool elevation the mean maximum biomass was 14.5 g/m^3 while at the upper pool elevation the mean maximum biomass was 7.8 g/m^3. The mean maximum biomass at the proposed pool elevation was 10.9 g/m^3.

127. Selective withdrawal was also evaluated at the upper pool elevation for 1975 (Figures 93-95). The phytoplankton concentrations were slightly less using selective withdrawal (6.1 g/m^3 of ALGAE 2 versus 7.8 g/m^3 with bottom withdrawal), but the anoxic period in the hypolimnion was extended. There were two periods of at least 30 days of consecutive anoxic conditions before the stratification was disrupted and oxygen remixed into the hypolimnion.

128. Destratification. The average year, 1971, was selected to

investigate the effects of destratification on water quality. Destratification was simulated by increasing the effective diffusion coefficient until complete mixing occurred throughout the year.

129. The bottom dissolved oxygen regime in the pool increased dramatically (Figure 96). At no time during the simulation period did the dissolved oxygen concentration decrease below 4 mg/ℓ. The dissolved oxygen in the bottom layer was usually equivalent to the surface layer concentrations (Figure 97).

130. Destratification did not alter the phasing or magnitude of the ALGAE 1 blooms but did effect the phasing of the ALGAE 2 blooms (Figures 98-99). Destratification resulted in a greater distinction among blooms with four blooms occurring during the same interval that three blooms occurred in without destratification. The magnitude of all the blooms was similar, however.

131. Increased minimum releases. The minimum low flow release was increased from 0.14 to 0.42 m^3/sec during 1976. This was the only year minimum flows were reached. There was no effect on the in-lake water quality with this increased flow (Figures 100-102).

132. Decreased maximum flow. The maximum flow release was reduced from 70 to 48.1 m^3/sec during the wet year, 1975. This change significantly altered water quality conditions within proposed Twin Valley Lake.

133. Hypolimnetic DO concentrations were lower for a longer period of the year (Figure 103). Excluding one spike of low DO, the oxygen regime increased after 16 August with the higher maximum release. With the decreased release, the oxygen regime does not significantly increase above 3 mg/ℓ until 13 October.

134. Both phytoplankton assemblages exhibited a significant response to the change in releases (Figures 104-105). The mean peak biomass of ALGAE 1 in the first bloom increased from 4.6 to 13.0 g/m^3 with decreased flow. Successive ALGAE 1 blooms were comparable under the two releases. The ALGAE 2 biomass also increased dramatically (Figure 105, note the change in the ordinate scale). The mean peak ALGAE 2 biomass increased from 10.9 to 40.9 g/m^3 with decreased flow.

The phasing of the blooms was similar under the two release schemes.

135. Selective withdrawal was also operated with the lower release to assess its impact on water quality. The following comparison is between bottom withdrawal and selective withdrawal operation at the lower release of 48.1 m^3/sec.

136. Selective withdrawal did result in an increase in the DO regime after 16 August although it was still slightly lower than the DO during the higher flow release (Figure 106).

137. Selective withdrawal also resulted in a reduction in phytoplankton blooms. The first ALGAE 1 bloom decreased from a mean maximum biomass value of 13.0 g/m^3 with bottom withdrawal to 9.5 g/m^3 with selective withdrawal (Figure 107). This was still substantially higher than 4.6 g/m^3 with the increased release. The mean peak biomass of ALGAE 2 was reduced to 17.3 g/m^3 with selective withdrawal (Figure 108). The comparative values with bottom withdrawal were 40.9 under the decreased flow regime versus 10.9 g/m^3 with the increased flow regime.

Discussion

138. Before the mathematical simulation results are discussed, it is important to realize that ecosystem modeling is as much an art as a science. No model is capable of predicting absolute values. When ecological systems are simplified to a few compartments and coefficients, as is the case with any model, mean coefficients may result in predictions that are off by as much as an order of magnitude from measured values. A broad range of coefficients is necessary to define all possible perturbations of the system. Monte Carlo simulations represent one way to estimate the extent of these perturbations. The Monte Carlo simulations also permit an expression of the variability inherent in the system. Associated with each measurement of a coefficient or update value is sampling error, experimental error, analytical error, and, in many instances, intrinsic variability of the organism. Monte Carlo simulations are one way of incorporating this variability. The results have been stated primarily in terms of the mean values or the

mean maximum. Associated with each of those mean values is a confidence interval indicating that there is a range of values that the variable may have at any given time. It is important in the following discussion and subsequent decisionmaking process to incorporate the magnitude of the range and not just the mean value. A large confidence interval about a mean indicates that many values are possible under varying conditions.

139. Consideration must also be given to the model assumptions and limitations (paragraph 63), the most important one being the one-dimensional assumption. The predictions are valid only in the deep part of the pool near the dam, not in the headwaters, coves, or embayments. The predictions are also valid only under aerobic conditions. It may be possible to predict when the DO goes out, but there is no mechanism in the model to account for the oxygen debt that builds up under anoxic conditions. Lastly, model predictions represent conditions after the transients in water quality from the initial filling have diminished. This may take 5 years or more.

Thermal simulations

140. The mathematical simulations indicated that proposed Twin Valley Lake would intermittently stratify from May through July. Periods of stratification extended from 5 to 45 days or more depending on hydrometeorological conditions. The downstream temperature objective was met with surface, bottom, and selective withdrawal. A selective withdrawal structure is not required to meet the natural temperature objective.

Water quality simulations

141. Dissolved oxygen. The simulation results indicated anoxic conditions could occur during intermittent periods of stratification. Considering the organic content of the soil and the watershed land use, this conclusion is reasonable. The model cannot presently simulate anaerobic processes; however, default algorithms are included to modify rate coefficients and reflect anoxic conditions, but the simulations do not include an oxygen debt. Since the thermal simulations indicated how rapidly proposed Twin Valley may mix, the requirement for additional

oxygen to satisfy the debt may not be a significant factor. It is also important to recall that the integration interval was 1 day so the results represent daily averages. If the oxygen debt were satisfied within several hours after mixing was initiated, the average daily condition predicted in the simulations may be representative of the prototype conditions.

142. The intermittent stratification of the pool may explain why the water quality was similar with the three withdrawal approaches. Selective withdrawal structures offer distinct advantages for regulating various water quality constituents in systems that strongly stratify (Fruh and Clay 1973). Bottom withdrawal has also been shown to improve water quality in reservoirs that experience anoxic hypolimnion (Dunst 1974; Moore 1976). These systems, however, have different morphometric and hydrodynamic properties than proposed Twin Valley Lake. Density interflows and underflows in strongly stratified systems can supply oxygen to the hypolimnion and reduce the supply of nutrients to the epilimnion (Dunst 1974). In proposed Twin Valley Lake, however, the inflow zone generally extended throughout the water column.

143. Increased DO at the lower pool elevation may be explained by the decreased residence time in the pool because of the smaller volume. Although the loadings per unit volume will be greater, the decreased residence time may prevent this material from exerting its ultimate demand before it passes through the pool. The increased pool elevation may have decreased the DO because the loadings were retained in the pool for a longer period and exerted a greater demand per unit mass. A similar situation probably occurred with the decreased maximum release. During 1975, a major storm event occurred around the end of June. With the lower release, the loading of organic matter associated with the storm event was retained in the pool for a longer period of time. The decay of this material resulted in extended anoxic conditions.

144. Algae. Predicted algal blooms ranged from 0.5 to 40.9 g/m^3. For comparative purposes, 0.7 g/m^3 is considered a visible bloom and 1.5 g/m^3 is considered a nuisance bloom. If the conversion factor developed by Spangler (1969) is used (i.e., 0.23 g/m^3 dry weight = 1 $\mu g/\ell$

chlorophyll a), then the magnitude of the predicted algal blooms in terms of chlorophyll a would be 2 to 89 μg/ℓ. Since the 40.9 g/m^3 represents blue-green algae, the chlorophyll a value is only half that for other phytoplankton species (Kalff and Knoechel 1978). This range of values is similar to the values listed for the NES lakes (Table 2).

145. Selective withdrawal or surface withdrawal, in general, resulted in lower phytoplankton concentrations. Since the pool was generally well mixed, the inflow was distributed through the water column regardless of the withdrawal scheme. However, because of the short residence time, selective and surface withdrawal may remove phytoplankton from the euphotic zone by discharging them through the outlet works. Berman and Pollingher (1974) noted that phytoplankton growing in a eutrophic lake had a doubling time of 8 to 30 days. Since the theoretical hydraulic residence time is 20 days, the phytoplankton may not be able to fully respond to the nutrient conditions before being removed from the reservoir. The selective withdrawal operation during the 1975 storm event may have short-circuited water through the euphotic zone further reducing the time for phytoplankton to respond to the increased nutrient loads. The increased phytoplankton concentrations during 1975, however, indicated a response to the nutrient loadings did occur.

146. One factor that has not been incorporated into the simulations is decreased light penetration due to turbidity and suspended solids. The USGS data indicated suspended solid concentrations of several hundred milligrams per litre in the stream. During and following storm events, these concentrations would be expected to increase and be transported into the pool. The extent of decreased light penetration and settling rate of these particles is unknown but it could significantly reduce the phytoplankton response to increased nutrients.

147. The phytoplankton algorithms do not consider silica limitation. Simulating diatom dynamics may, therefore, be in error by omitting silicon uptake. Recently, Kalff and Knoechel (1978) provided a relationship between maximum observed diatom biomass and the silicon concentrations for lakes in which silicon depletion occurs. Using this relationship and an average silicon concentration of 4.0 mg/ℓ,

the maximum diatom biomass supported by this silicon concentration would be 21.3 g/m^3. Since this concentration of ALGAE 1 was never attained in the simulations, silicon limitation may not be an important consideration in phytoplankton magnitude and succession in proposed Twin Valley Lake.

148. Destratification was simulated by increasing the effective diffusion coefficient in the model. This is probably comparable to the input of external mechanical energy to mix the lake. A Garton pump or other source of mechanical energy used in destratification simply overcomes the buoyant forces generated from the input of solar energy. Increasing the effective diffusion coefficient provides a similar energy input.

149. Fecal coliforms. Simulated fecal coliforms did not exceed the State standard of 200 colonies/100 ml. It was only after the inflow concentrations were increased by a factor of one hundred that the simulations showed violations of State standards. During these periods, the inflowing counts were approximately 5000/100 ml or one order of magnitude larger than the maximum value measured (Table 1). Some violation of the standards will probably occur in the headwaters whenever the inflowing counts are greater than 200 colonies/100 ml because the inflows are not instantaneously dispersed throughout the pool as assumed by the model.

PART VI: LOADING ANALYSES

150. Cultural eutrophication is generally accepted as an acceleration of the natural eutrophication process due to man's activities in the watershed. The activities result in excess nutrient discharges into surface waters. These nutrient discharges occur from both point (e.g., sewage and industrial outfalls) and nonpoint (agricultural and urban runoff) sources in the watershed. One of the earliest studies attempting to quantify the trophic status of the lentic environments based on external nutrient loads was conducted by Vollenweider (1968). Since 1968, numerous studies have been conducted to modify and improve the nutrient loading concept (Shannon and Brezonik 1972; Dillon 1974, 1975; Dillon and Rigler 1975; Kirchner and Dillon 1975; Larsen and Mercier 1976; Vollenweider 1975, 1976; Carlson 1977; Kalff and Knoechel 1978). Nutrient loading models or the loading concept has also been extended to predict water clarity and average chlorophyll concentrations (Carlson 1977; Dillon and Rigler 1974; Vollenweider 1976; Jones and Bachmann 1976) and the probability of developing anoxic conditions (Reckhow 1978).

151. Since the nutrient loading concept is predicated primarily on the external nutrient load and various morphometric characteristics of the impoundment rather than measured in-lake quantities, it may be applied for predictions of the trophic status expected to occur in proposed impoundments. Several of the nutrient loading models, therefore, were investigated to project the trophic status in the proposed Twin Valley Lake.

Assumptions and Limitations

152. It is important to consider the assumptions of the nutrient loading models in assessing the trophic status of an impoundment before any definitive conclusions can be drawn. While the equations are simple and relatively easy to apply, it should not be inferred that conclusions concerning the trophic status can be directly and simply drawn from

the results. The relative position of a proposed or existing impoundment on a loading plot, specifically in the eutrophic zone, does not necessarily indicate the magnitude or degree of eutrophication. While the nutrient loading concept is a reasonable approach for demarcation of the general trophic status, e.g., oligotrophic, mesotrophic, or eutrophic, there are many other factors that affect the trophic state of a reservoir such as the zone of withdrawal, stratification, alkalinity, cation ratios, and others.

153. Several assumptions have previously been stated by Dillon (1974):

a. Loading, flushing, and sedimentation rates are assumed to be in steady state.

b. Stratification or nonuniform mixing is ignored; the system is considered to be a well-mixed reactor.

c. The concentration of a substance in the outflow is equated to the mean concentration in the lake.

154. Several corollary assumptions follow:

a. The substance under consideration is limiting.

b. There is a direct relation between loadings and biomass.

c. A reduction in loadings will result in a concomitant reduction in indicators of eutrophication.

155. While the applicability and validity of the assumptions will be discussed later, it is important to refer to the assumptions during the subsequent calculations of trophic status for proposed Twin Valley Lake.

Application of Nutrient Loading Models

156. Water quality samples collected at the Twin Valley Gage on the Wild Rice River by the USGS from 1975-77 were used to develop regression equations for total phosphorus and nitrogen as a function of flow. These equations represented one approach for generating annual nutrient loads to the proposed project based on daily flow records. Two equational forms were used to generate annual nutrient loads. The first equation was simply the product of mean annual flow and mean annual

nutrient concentration. The second equational form was a power relationship between nutrient mass and flow. The regression equations for phosphorus and nitrogen mass had coefficients of determination, R^2, of 0.92, and 0.75, respectively. Flows for the three study years (1971, 1975, and 1976) were used to generate nutrient loads representative of average, wet, and dry years, respectively. The equations and the generated nutrient loads are listed in Table 7. Critical phosphorus loads representing the transition points between dangerous and excessive loads were calculated for all three study years using the equations of Vollenweider (1976). These results are shown in Table 8. Optimal N/P ratios of 8 to 12:1 (Chiaudani and Vighi 1974) were used to calculate critical nitrogen loads corresponding to the critical phosphorus loads. If actual phosphorus loads exceed the critical phosphorus loads but actual nitrogen loads are less than the critical nitrogen loads, nitrogen limitation is indicated. Table 8 indicates that actual nitrogen loads are, indeed, less than critical nitrogen loads.

157. While arguments can be proposed for the coprecipitation of phosphorus with $CaCO_3$ and the addition of nitrogen through phytoplankton nitrogen fixation, their quantification is speculative. Various phosphorus loading analyses are included in Appendix F as an academic exercise. Under known phosphorus limitation, nutrient loading models provide only a relative estimate of eutrophication potential, not absolute predictions, and several of the assumptions required to apply many phosphorus loading models are already violated. First, loading calculations indicate that nitrogen, not phosphorus, is the limiting nutrient, therefore phosphorus loading models are not applicable. This conclusion is corroborated by the NES studies on surrounding lakes within a 200-km radius of Twin Valley. Many of the bioassay tests conducted on these lakes indicated nitrogen limitation. Second, the steady-state assumption is known to be violated for the proposed Twin Valley Lake. Theoretical hydraulic residence times vary from 4 to 945 days. Nutrient loadings to the proposed project are not uniform throughout the year. Third, various chemical and biological reactions such as coprecipitation of phosphorus or nitrogen fixation may not result in a

direct relation among loadings and biomass. As evidenced by the current research activities in the area of loading calculations (Carlson 1977; Dillon 1974, 1975; Dillon and Rigler 1974, 1975; Kirchner and Dillon 1975; Larsen and Mercier 1976; Shannon and Brezonik 1972; Vollenweider 1975, 1976), there are many factors such as phosphorus retention rate, hydraulic residence time, flushing rate, basin morphometry, alkalinity, and others that affect the loading, availability, and uptake of nutrients in the eutrophication process. The loading concept originally proposed by Vollenweider was not intended to predict absolute degrees of eutrophy but rather to permit a relative comparison of loadings to estimate the potential for eutrophication (Vollenweider 1968).

158. One use of phosphorus loading values that is not dependent on an assumption of phosphorus limitation is the prediction of anoxic conditions in a lake. Reckhow (1978) developed a classification function based on phosphorus loading, mean depth, and theoretical hydraulic residence time that permits a categorization of lakes according to oxic-anoxic conditions in the hypolimnion. The value of the classification function is that it permits a calculation of the probability that anoxic conditions will occur. The function was developed from data collected on EPA-NES lakes during 1972. These data included the Minnesota lakes sampled during the NES program. The classification and discriminant functions used in the analyses are listed in Table 9. In general, the geomorphological variables, mean depth and volume, had greater discriminating power than the loading variable.

159. The classification and discriminant functions were used to generate probabilities of anoxic conditions occurring in proposed Twin Valley Lake (Table 10). The probability of anoxic conditions developing in proposed Twin Valley Lake ranged from 95 to 99 percent based on the loading analyses for the study years with a mean of 97 percent for all years.

160. Three factors characteristic of the NES lakes used by Reckhow (1978) had the following ranges:

mean depth > 3.0 m

residence time > 0.25 yr

$$1.0 < \frac{\text{mean depth}}{\text{residence time}} > 50.0 \text{ m/yr}$$

The proposed Twin Valley Lake has a mean depth of 4.2 m, but the residence time is a fifth to a tenth of the NES lakes. The quotient, mean depth/residence time, is, therefore, also outside the range of the NES lakes. It is not known if these factors influence the prediction of anoxic conditions. Considering the watershed land use, phosphorus loadings, and probable organic loadings, the probability of at least some period of zero DO appears reasonable.

PART VII: DISCUSSION OF PREDICTIONS AND CRITERIA

161. In the studies described in the preceding sections, different techniques were used to predict the water quality and trophic status of proposed Twin Valley Lake. The predictions will be discussed, compared, and related to appropriate water quality criteria in this section. Thermal stratification, DO, algae and macrophytes, trophic state, fecal coliforms, reservoir clearing and filling, and management alternatives are discussed.

Thermal Stratification

162. In the Final Environmental Impact Statement (U. S. Army Engineer District, St. Paul 1975a), an equation developed by R. A. Ragotskie was used to determine that proposed Twin Valley Lake would thermally stratify. This equation is compared with data collected by Ford (1976) in Figure 109. Ragotskie's equation predicted a shallower thermocline than the data support. Based on Figure 109, a thermocline depth of over 9 m is predicted for proposed Twin Valley Lake. Since this depth is near the maximum depth of the project (i.e., 9.5 m), thermal stratification is not expected. This conclusion is also supported by data on surrounding lakes. All of the lakes that were morphometrically similar to proposed Twin Valley Lake stratified only weakly, if at all. Less stratification is expected in proposed Twin Valley Lake because the theoretical hydraulic residence time is less than the surrounding lakes (paragraph 32).

163. Based on the mathematical simulations, intermittent stratification is expected from May through July. The periods of stratification may last up to 45 days or longer. These predictions are expected to be conservative (paragraph 91).

164. The sheltering effect of the surrounding terrain is not expected to be significant. For the months June through October, the prevailing wind direction is south-southeast. Since this direction is perpendicular to the major axis of the lake, the sheltering effect

should be the greatest under these conditions. Based on laboratory and field data, the sheltering effect is approximately eight times the vertical relief (Ford 1976). Assuming the surrounding relief to be on the order of 20 to 25 m and a typical fetch to be 600 m, then the sheltered area would extend 160 to 200 m into the lake. Less than one third of the lake surface area would be sheltered from the wind and in-lake mixing should not be affected.

165. Proposed Twin Valley Lake is expected to be intermittently stratified during early summer. No hypolimnion, isolated from the epilimnion, will exist in the classical sense. Thermal gradients near the sediment will not prevent the diffusion of material into the epilimnion.

Dissolved Oxygen

166. The Minnesota standard for DO in proposed Twin Valley Lake is not less than 6 mg/ℓ from 1 April through 31 May and not less than 5 mg/ℓ at other times (MPCA 1973). The hypolimnion is excluded. Proposed Twin Valley Lake is not expected to violate this standard.

167. Based on the mathematical simulations and Reckhow's classification function, proposed Twin Valley Lake is expected to go anoxic during periods of stratification. Data from surrounding, morphometrically similar lakes do not support this conclusion. Since these lakes were sampled three times or less, periods of anoxia could have been missed. The mathematical simulations and laboratory experiments (Appendix B) predicted it would take 5 to 15 days, after stratification, for anoxic conditions to develop. The duration of anoxia will depend on hydrometeorological conditions and the resulting thermal stratification. Periods of up to 100 days are possible but not likely. Anoxia will be limited to the bottom 2 or 3 m of the lake which comprises less than 15 percent of the lake volume.

168. Once anaerobic conditions have developed, inorganic carbon, ammonium, and orthophosphate will accumulate in the anoxic zone. Any period of mixing, however slight, will release these nutrients to the

epilimnion where they will be available for algal growth.

169. The laboratory experiments also indicated that sulfide production is possible. Assuming the laboratory experiments can be extrapolated to the field, then it may take as little as 10 to 15 days for sulfide to start to accumulate.

Algae and Macrophytes

170. The types of algae found in the surrounding lakes and expected to occur in proposed Twin Valley Lake are summarized in Table 5. Diatoms and green algae are expected to dominate in the spring and fall while blue-green algae are expected in the summer. Many of the blue-green algae in Table 5 will accumulate on the water surface during large blooms. Since the prevailing wind direction during the summer months is south-southeast, algae would probably accumulate on the north side of the lake.

171. The algal concentrations found in surrounding impoundments ranged from 1 to 130 μg/ℓ of chlorophyll *a* (Table 2). This is consistent with the mathematical model predictions of 2 to 89 μg/ℓ of chlorophyll *a* for blooms. Chlorophyll *a* concentrations of the order of 20 to 50 μg/ℓ are expected in proposed Twin Valley Lake. Larger concentrations of algae are expected in the headwater regions and coves. The mathematical model is not capable of simulating this effect because of the one-dimensional assumption.

172. Conditions within the proposed lake are also expected to be favorable for macrophyte colonization and growth (Appendix C). On the basis of light penetration alone and assuming no other factors are limiting, a maximum of approximately 46 percent of the lake surface is potentially colonizable. Although most of the macrophytes are expected to occur in the headwaters region, colonization is also probable in the littoral zones near the recreational areas.

Trophic State

173. The trophic state of an impoundment refers to the degree of

nutrient enrichment. Lakes are usually classified as oligotrophic, mesotrophic, or eutrophic in the order of increasing enrichment. The problem with this classification is that it is subjective, and definitions vary from one part of the country to another.

174. The Great Lakes Group (1976) recommended that concentrations of 7 to 8 μg/ℓ of chlorophyll a separate mesotrophic from eutrophic lakes, while the National Eutrophication Survey (Gakstatter et al. 1975) recommended 10 μg/ℓ. Using these criteria, the modeling predictions and the data from surrounding impoundments indicate that proposed Twin Valley Lake would be eutrophic.

175. Miller et al. (1978) considered waters containing 0.015 mg/ℓ bioavailable phosphorus and 0.165 mg/ℓ bioavailable nitrogen to be eutrophic. To control cultural eutrophication, EPA (1976) recommended that phosphorus concentrations should not exceed 0.05 mg/ℓ in any stream entering a reservoir and in-lake concentrations should not exceed 0.025 mg/ℓ. Mean phosphorus and TSIN concentrations in the Wild Rice River were 0.057 mg/ℓ and 0.11 mg/ℓ, respectively (Table 1). Based on phosphorus concentrations, proposed Twin Valley Lake is expected to be eutrophic.

Fecal Coliforms

176. The Minnesota standard for fecal coliforms is 200 Most Probable Number per 100 ml as a monthly geometric mean (MPCA 1973). The EPA recommended criterion for body contact recreation is 200 colonies/100 ml based on a logarithmic mean of a minimum of five samples in 30 days (EPA 1976). The value of 200 colonies/100 ml will be used here.

177. The model simulations predicted no problems meeting these criteria. However, since the model is one-dimensional and not able to simulate longitudinal variations and since some of the inflow counts were over 200 colonies/100 ml (Table 1), it is expected that periodic violations may occur in the headwater regions.

Reservoir Clearing and Filling

178. During the first 6 to 8 years after project filling, a reservoir undergoes dynamic biological and chemical changes. Many of these are directly or indirectly associated with decaying organic matter which was inundated upon filling. To minimize the impact of reservoir filling on water quality, laboratory studies, using soil samples from the project area, were undertaken (Appendix B).

179. The laboratory studies indicated that the initial oxygen demand resulting from flooding the soil could be reduced by clearing the land of vegetation. Removal of the entire A horizon would result in a more significant decrease.

180. Readily soluble and leachable components could be removed from the proposed reservoir by a series of fillings and flushing prior to the initial filling of the reservoir. Assuming an average year (Figure 6), the proposed reservoir could be filled and flushed three times from April to June with the final filling occurring during July. Efficient decomposition of organic matter could be promoted by a period of slow incremental filling to keep the reservoir shallow, thereby avoiding thermal stratification.

Management Alternatives

181. Several alternative operational approaches were evaluated to assess their impact on water quality and project purposes. These approaches included bottom and surface withdrawal, lower and higher pool elevations, increased minimum and decreased maximum releases, and destratification.

Withdrawal

182. The mathematical simulations indicated that proposed Twin Valley Lake could be operated to meet the downstream natural stream temperature objective with bottom, surface, or selective withdrawal. In addition, no differences were observed in in-lake water quality with the three withdrawal schemes. Since the lake does not strongly stratify,

selective withdrawal offers no distinct advantages over bottom or surface withdrawal. Bottom withdrawal is therefore recommended.

183. If a selective withdrawal structure is considered necessary to provide flexibility in structure operation and maintenance, the original design (Figure 28) should be modified. Consideration should be given to:

a. Adding a "piggyback" gate to the flood control gate to release small flows (i.e., less than 1.4 m^3/sec) from the bottom of the pool.

b. Using a single wet well. Since the proposed lake is not expected to stratify strongly and since the withdrawal zone usually extends through the entire water columm, blending between ports is not a major consideration. However, blending is still possible in a single well system because blockage due to density stratification in the wet well is not expected to be a problem.

c. Reducing the size of the water quality ports for a maximum release of approximately 4.3 m^3/sec.

Pool elevation

184. Pool elevations were raised and lowered by 1.5 m to determine the effect of pool elevation and residence time on water quality. The simulations for the higher pool elevation can be interpreted to correspond to two different project operations. In the first operation, the pool is held constant at the higher elevation all year. This operational scheme would adversely affect flood control operations and benefits. In the second operation, dual storage operation is assumed. The pool is raised from its winter conservation level to the summer conservation pool level during the spring flood.

185. Generally, the simulations indicated that the lower the pool the better the DO and the worse the phytoplankton. Lowering the pool elevation is not recommended because recreation would be severely affected by increased phytoplankton and reduced surface area. Macrophytes would also be a problem. Dual storage would be the only effective way to raise the pool. The flood control benefits would be retained along with a larger pool for recreational purposes. Although the phytoplankton decreased slightly at the upper pool level, the duration and extent of anaerobic conditions increased. The slight decrease in

phytoplankton is probably not worth the increased period of anaerobic conditions. The original pool elevation (324 m msl) is recommended.

Releases

186. Increasing the minimum release from 0.14 to 0.42 m^3/sec in 1976 had no effect on in-lake water quality. Conditions would probably have improved downstream, however. Since the routings and simulations were for only 1 year, the effects of a prolonged drought on water quality could not be determined.

187. Decreasing the maximum release to 48 m^3/sec in 1975 significantly altered the water quality of proposed Twin Valley Lake for the worse. Since 1975 was the second wettest year on record and since the flood occurring near the end of June was rare, the realization of these conditions will also be rare. The simulations do, however, indicate that whenever flood waters are stored, water quality would be adversely affected.

Destratification

188. With destratification the lake remained aerobic all year. Although phytoplankton blooms were predicted to increase, the precise effects of mechanically mixing the lake are unknown.

PART VIII: CONCLUSIONS

189. Conclusions based on this study are as follows:

a. Proposed Twin Valley Lake will probably stratify intermittently from May through July.

b. State standards for DO will not be violated. The bottom waters will go anoxic 5 to 10 days after stratification forms. Hydrogen sulfide production is possible after 10 to 15 days of stratification.

c. Blooms of blue-green algae are possible throughout the summer. The magnitude of these blooms will be similar to those of surrounding lakes. During large blooms, surface accumulation is probable.

d. State standards for fecal coliforms will probably not be violated at the proposed recreation areas.

e. Proposed Twin Valley Lake will be eutrophic.

f. Both bottom and selective withdrawal met the downstream temperature objective. The in-lake water quality was also similar for both withdrawal schemes. Bottom withdrawal is recommended.

g. Prior to filling, removal of all vegetation will probably reduce the initial oxygen demand.

h. A series of fillings and flushing followed by a slow incremental filling will probably improve water quality during initial years.

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Table 1

Summary of Water Quality Data Taken at the USGS Gage at Twin Valley

Variable	1975 Mean	1976 Mean	1977 Mean	Mean	Minimum Value	Maximum Value
Dissolved oxygen (DO), mg/ℓ	9.0(5)*	9.6(29)	9.4(21)	9.5(55)	6.1	13.4
Biochemical oxygen demand (BOD5), mg/ℓ	-	2.6 (18)	8.5(18)	5.5(36)	1.0	23.0
Total phosphorus (P), mg/ℓ	0.079(7)	0.056(31)	0.051(24)	0.057(62)	0.01	0.24
PO_4-P, mg/ℓ	-	0.023(29)	0.027(20)	0.024(49)	0.01	0.10
Total soluble inorganic nitrogen (TSIN), mg/ℓ	0.23(6)	0.074(27)	0.119(24)	0.11(57)	0.0	0.64
NO_3-N, mg/ℓ	0.17(6)	0.04(28)	0.072(24)	0.066(58)	0.0	0.64
NH_4-N, mg/ℓ	0.065(6)	0.03(29)	0.048(24)	0.041(59)	0.01	0.33
NO_2-N, mg/ℓ	0.0(6)	0.0(28)	0.0(24)	0.0(58)	0.0	0.0
SiO_2, mg/ℓ	14.9(7)	12.6(28)	13.6(24)	13.3(59)	1.5	22.0
Alkalinity, mg/ℓ as $CaCO_3$	228(7)	251(30)	249(24)	248(61)	150	347
pH	8.0(7)	8.2(30)	8.3(24)	8.2(61)	7.3	8.8

(Continued)

* Number in parentheses is number of values used to calculate mean.

Table 1 (Concluded)

Variable	1975 Mean	1976 Mean	1977 Mean	Mean	Minimum Value	Maximum Value
Specific conductance, µmhos/cm	402(43)	549(173)	553(254)	537(461)	254	844
Total dissolved solids (TDS), mg/ℓ	303(7)	307(26)	327(24)	315(57)	231	369
Fecal coliforms, colonies/100 ml	31(4)	63(25)	-	58(29)	6	390
Total streptococci, colonies/100 ml	31(5)	65(24)	-	59(29)	2	248

Table 2

Summary of NES Data

Lake Name	Storet No.	County	Distance, km, and Direction from Twin Valley	Lake Type*	Surface Area, ha	Mean Depth, m	Maximum Depth, m	Volume, 10^6 m^3	Mean Hydraulic Residence Time, yr	Conductivity Range, umhos/cm	Secchi Disc Depth Range, m	Stratification, °C*	Minimum DO, mg/ℓ*	pH Range	Alkalinity Range, mg/ℓ $CaCO_3$	N:P Ratio	Limiting Nutrient*	Total Annual P Load, $g/m^2/yr$	Accumulated Annual P Load, $g/m^2/yr$	Total Annual N Load, $g/m^2/yr$	Accumulated Annual N Load, $g/m^2/yr$	Chlorophyll a Range, ug/ℓ*	Trophic State*
											Minnesota Lakes												
Andrusia	2700	Beltrami	122 ENE	N	611	7.9	18.3	48.4	0.13	240-345	1.1-1.5	11.6	0.3	7.0-8.5	136-183	8:1-40:1	U	4.02	1.47	59.1	6.9	11.0-17.4J	E
Badger	2704	Polk	50 NNE	N	143	1.2	3.4	1.75	0.30	450-485	1.3-2.4	--	10.2	8.4-8.5	183-185	7:1-17:1	U	0.63	--	10.6	0.03	0.9-3.6J	E
Bartlett	2705	Koochi-ching	167 ENE	N	123	2.6	4.6	3.18	1.9	220-258	0.3-0.8	0.5	7.2	8.3-8.9	93-121	8:1-16.1	U	0.37	0.21	3.4	0.6	25.8-63.9J	E
Bemidji	2701	Beltrami	104 ENE	N	2,598	9.8	23.2	253	0.73	290-380	1.8-2.5	9.8	0.2	7.3-8.4	149-183	9:1	N	0.44	0.16	13.4	3.3	6.6-17.0J	E
Big Stone	2709	Big Stone	218 S	N	5,103	3.4	4.9	171	1.7	730-980	0.4-2.7	0.5	6.0	7.8-8.7	124-196	4:1	U	0.31	--	4.3	0.1	0.6-54J	E
Birch	2710	Cass	135 ESE	N	519	3.1	13.7	15.8	--	150-265	1.4-4.3	12.0	0.3	6.5-8.7	88-129	2.5:1-24:1	N,P	--	--	--	--	1.5-11.7J	E
Blackduck	2711	Beltrami	140 ENE	N	1,110	4.5	11.3	50	4.2	240-260	0.9-2.3	1.6	6.3	7.2-8.7	121-131	8:1-13:1	P	0.14	0.08	2.6	1.2	9.8-28.7J	E
Blackhoof	2712	Crow Wing	193 SE	N	74	4.4	9.1	3.27	0.70	199-275	1.4-1.8	15.5	0.0	7.2-8.6	94-122	5:1-14:1	N	1.22	1.03	11.8	4.9	5.7-19.8J	E
Carlos	27B9	Douglas	160 SSE	N	1,020	13.1	49.7	134	3.7	320-420	1.8-3.7	14.0	2.0	8.0-8.6	161-199	17:1	P	0.14	0.08	6.7	3.0	3.2-6.4J	M
Cass	2715	Beltrami & Cass	130 ENE	N	6,312	7.6	36.6	481	0.86	248-330	1.1-3.1	14.9	0.04	7.2-8.5	132-175	7:1-28:1	U	0.35	0.22	8.8	1.6	3.2-16.4J	M
Darling	27B4	Douglas	166 SSE	N	386	6.2	18.9	23.9	0.87	340-475	2.1-2.8	10.7	0.0	7.3-8.5	190-218	8:1-21.1	U	0.19	0.05	9.1	0.6	8.6-18.2J	M-E
Gull (South Basin)	2737	Cass & Crow Wing	175 SE	N	3,861	9.1	34.8	353	2.9	187-220	2.0-3.0	13.0	0.0	7.0-8.6	93-123	6:1-16.1	N	0.10	0.04	3.7	1.4	4.8-19.1J	E
Leech	2746	Cass	124 E	N	45,326	4.7	45.7	2141	5.2	239-310	1.7-3.0	10.2	1.0	7.3-8.4	129-154	5:1-14:1	N	0.04	0.02	2.4	1.2	3.2-8.7J	M
Le Homme Dieu	27B5	Douglas	166 SSE	N	706	6.4	25.9	45.2	7.9	290-370	1.2-3.7	13.0	0.0	7.3-8.6	142-216	10:1-30:1	P	0.12	0.10	3.4	2.2	4.3-28.8J	E
Little	2748	Grant	140 S	N	28	--	1.2	--	--	325-825	0.4-2.0	--	7.1	9.0-9.4	190-296	1:1-8:1	N	--	--	--	--	5.8-130	Hy-per E
Malmedal	2752	Pope	190 SSE	N	79	1.8	2.4	1.4	1.3	520-600	0.3-0.4	0.1	5.4	8.0-8.7	155-193	4:1-19:1	P	--	--	--	--	9.7-63.5J	E
Minnewaska	2761	Pope	200 SSE	N	2,877	6.0	9.8	172	12.7	580-750	1.0-2.5	7.0	4.4	8.2-8.7	180-250	4:1-14:1	U	0.15	0.13	2.3	1.8	3.4-10.9J	E
Rabbit	2771	Crow Wing	197 SE	N	215	7.2	15.2	15.5	--	255-300	1.5-3.7	14.0	0.0	7.2-8.6	102-131	4:1-24:1	U	--	--	--	--	2.6-11.4J	E
Trace	2792	Todd	200 SE	N	112	--	--	--	--	--	0.9-1.2	--	8.7	8.1-9.6	154-206	--	--	--	--	--	--	1.7-37.8J	E
Winona	27A1	Douglas	170 SSE	N	78	2.4	13.4	1.0	1.5	330-420	0.4-0.6	6.2	7.6	8.3-8.7	118-164	6:1-14:1	N	1.65	1.58	7.3	6.2	14.9-88.7J	E
Wolf	27A2	Beltrami, Hubbard	120 ENE	N	425	8.5	17.7	36.3	0.10	248-355	0.8-1.5	12.9	0.4	7.2-8.7	137-169	3:1-12:1	N	6.43	0.80	60.3	--	12.7-25.8J	E

(Continued)

* Entries in these columns are defined in the text.

Table 2 (Concluded)

Lake Name	Storet No.	County	Distance, km, and Direction from Twin Valley	Lake Type*	Surface Area, ha	Mean Depth, m	Maximum Depth, m	Volume, 10^6 m^3	Mean Hydraulic Residence Time, yr	Conductivity Range, umhos/cm	Secchi Disc Depth Range, m	Stratification, °C*	Minimum DO, mg/ℓ*	pH Range	Alkalinity Range, mg/ℓ $CaCO_3$	N:P Ratio	Limiting Nutrient*	Total Annual P Load, $g/m^2/yr$	Accumulated Annual P Load, $g/m^2/yr$	Total Annual N Load, $g/m^2/yr$	Accumulated Annual N Load, $g/m^2/yr$	Chlorophyll *a* Range, ug/ℓ*	Trophic State*
North Dakota Lakes																							
Astabula	3801	Barnes, Griggs	140 W	I	2,198	4.0	13.7	87.9	0.83	265-867	0.3-1.2	0.8	4.4	7.6-9.0	104-515	0.5:1-6:1	N	1.46	0.28	13.8	2.7	5.4-60.9	E
Jamestown River	3806	Stutsman	205 W	I	486	7.5	9.2	36.7	1.6	417-630	0.6-3.4	0.5	6.2	8.0-8.7	154-266	2:1-16:1	N,P	0.92	1.55	10.4	3.2	2.3-74.2	E
La Moure	3807	LaMoure	200 SW	I	200	5.0	12.2	9.96	1.9	333-733	0.8-3.7	8.4	0.0	7.6-8.4	128-515	1:1-4:1	N	1.35	0.56	5.2	0.7	2.6-39.2	E
Matejcek	3808	Walsh	180 NW	I	53	4.2	12.2	2.23	0.65	268-522	0.4-0.9	12.1	0.6	7.3-8.7	108-204	3:1-7:1	N	1.51	0.55	14.1	0.2	1.9-3.8	E
Spritwood	3813	Stutsman	200 W	N	167	8.8	16.2	14.7	--	1720-2900	0.9-3.7	13.4	0.0	8.5-9.0	255-730	3:1-6:1	N	--	--	--	--	2.5-84.3	E
Whitman	3815	Nelson, Walsh	180 NW	I	58	2.7	7.7	1.57	0.7	263-520	0.5-0.9	2.1	5.8	7.6-8.9	96-238	0.5:1-8:1	N	0.83	--	9.2	--	3.5-77.2	E

Table 3

Phytoplankton Genera Found in Surrounding Impoundments Within a 200 km Radius

Cyanophyta (Blue-Green)	Chlorophyta (Green)	Chrysophyta (Yellow-Brown	Pyrrophyta (Dinoflagellates)
Anabaena sp.	*Chlamydomonas* sp.	*Achnanthes* sp.	*Ceratium* sp.
Aphanizomenon sp.	*Closterium* sp.	*Asterionella* sp.	*Chroomonas* sp.
Aphanocapsa sp.	*Crucigenia* sp.	*Cocconeis* sp.	*Cryptomonas* sp.
Aphanothece sp.	*Dictyosphaerium* sp.	*Cyclotella* sp.	*Gymnodinium* sp.
Chroococcus sp.	*Elaktothrix* sp.	*Dinobryon* sp.	*Rhodomonas* sp.
Coelosphaerium sp.	*Kirchneriella* sp.	*Fragilaria* sp.	
Dactylococcopsis sp.	*Oocystis* sp.	*Gomphonema* sp.	
Gleotrichia sp.	*Pandorina* sp.	*Melosira* sp.	
Gomphosphaeria sp.	*Pediastrum* sp.	*Navicula* sp.	
Lyngbya sp.	*Scenedesmus* sp.	*Nitzschia* sp.	
Marssoniella sp.	*Sphaerocystis* sp.	*Rhizosolenia* sp.	
Merismopedia sp.	*Tetraedron* sp.	*Stephanodiscus* sp.	
Microcystis sp.		*Surirella* sp.	
Oscillatoria sp.		*Synedra* sp.	
		Synura sp.	
		Tabellaria sp.	

Table 4

Summary of the Major Modifications Made by WES to WQRRS Reservoir Model

Modification	Description
1	The model's representation of a reservoir was changed from a series of fixed horizontal layers to a series of variable layers to reduce numerical dispersion and ensure accurate mass balances. The thickness and volume of a layer depend solely on inflows and outflows to that layer.
2	The BOD compartment was replaced by a dissolved organic (DOR) compartment because BOD values may include effects of nitrification, decay of particulate organic matter, and algal respiration, which are included in other parts of the reservoir model.
3	The diffusion coefficient was modified to be a function of wind speed so that effects of wind mixing could be simulated.
4	The basis for determining reservoir stratification stability was changed from temperature to density differences so that inverse stratifications at temperatures below $4^{o}C$ could be simulated.
5	The inflow algorithm was modified so that under isothermal conditions, the inflow could be placed on the surface or at the bottom when the inflow differed in density from the isothermal reservoir by more than a specified amount.
6	A predator was added to the fish compartment, and the planktivore was modified to feed on detritus also.
7	The calculation of production was corrected to correspond to net primary production above the 1 percent light level.
8	The fraction of solar radiation absorbed in the surface layer was made variable.
9	The zooplankton compartment was modified so that they feed on both detritus and algae.
10	The model was modified so it could be run in a Monte Carlo mode, with specified distributions for coefficients and updates.

Table 5

Phytoplankton Genera Comprising 99 Percent of Standing Crop*
in Each Taxonomic Division in Surrounding Impoundments

Cyanophyta		Chlorophyta		Chrysophyta		Pyrrophyta	
Species	%	Species	%	Species	%	Species	%
Anabaena sp.	19	*Ankistrodesmus* sp.	12	*Asterionella* sp.	3	*Chroomonas* sp.	33
Aphanizomenon sp.	4	*Chlamydomonas* sp.	11	*Cyclotella* sp.	15	*Cryptomonas* sp.	50
Aphanocapsa sp.	1	*Closterium* sp.	6	*Dinobryon* sp.	29	*Rhodomonas* sp.	16
Chroococcus sp.	8	*Crucigenia* sp.	2	*Fragilaria* sp.	14		
Lyngbya sp.	32	*Dictyosphaerium* sp.	25	*Melosira* sp.	30		
Marssoniella sp.	2	*Kirchneriella* sp.	12	*Stephanodiscus* sp.	1		
Merismopedia sp.	4	*Oocystis* sp.	3	*Synedra* sp.	6		
Microcystis sp.	22	*Pediastrum* sp.	4	*Tabellaria* sp.	1		
Oscillatoria sp.	7	*Scenedesmus* sp.	24				

* Expressed in number per millilitre.

Table 6

Distribution of Coefficients and Updates

Variable	1971			1975			1976		
	$\bar{X}$±S.D.	Range	Type	$\bar{X}$±S.D.	Range	Type	$\bar{X}$±S.D.	Range	Type
ALGAE1 Growth Rate	1.42±1.43	0.1-3.8	Normal	1.42±1.43	0.1-3.8	Normal	1.42±1.43	0.1-3.8	Normal
ALGAE2 Growth Rate	1.61±1.84	0.1-3.6	Rotated	1.61±1.84	0.1-3.6	Rotated	1.61±1.84	0.1-3.6	Rotated
Zooplankton Growth Rate	0.505±0.05	0.4-0.6	Normal	0.505±0.05	0.4-0.6	Normal	0.505±0.05	0.4-0.6	Normal
ALGAE Respiration Rate	0.17±0.075	0.02-0.32		0.17±0.075	0.02-0.32		0.17±0.075	0.02-0.32	
Zooplankton Respiration Rate	0.2±0.04	0.12-0.28		0.2±0.04	0.12-0.28		0.2±0.04	0.12-0.28	
Zooplankton Mortality Rate	0.005±0.001	0.0031-0.007		0.005±0.001	0.0031-0.007		0.005±0.001	0.0031-0.007	
ALGAE1 P Half-Saturation Constant	0.003±0.001	0.001-0.006	Rotated	0.003±0.001	0.001-0.006	Rotated	0.003±0.001	0.001-0.006	Rotated
ALGAE2 P Half-Saturation Constant	0.006±0.002	0.001-0.011		0.006±0.002	0.001-0.011		0.006±0.002	0.001-0.011	
ALGAE1 N Half-Saturation Constant	0.014±0.014	0.001-0.04		0.014±0.014	0.001-0.04		0.014±0.014	0.001-0.04	
ALGAE2 N Half-Saturation Constant	0.010±0.012	0.001-0.04		0.010±0.012	0.001-0.04		0.010±0.012	0.001-0.04	
ALGAE1 CO_2 Half-Saturation Constant		0.05-0.15	Uniform		0.05-0.15	Uniform		0.05-0.15	Uniform
ALGAE2 CO_2 Half-Saturation Constant		0.05-0.15			0.05-0.15			0.05-0.15	
ALGAE1 Light Half-Saturation Constant		4-8			4-8			4-8	
ALGAE2 Light Half-Saturation Constant		2-6			2-6			2-6	
ALGAE1 Settling Rate		0.08-0.5			0.08-0.5			0.08-0.5	
ALGAE2 Settling Rate		0.0-0.2			0.0-0.2			0.0-0.2	
Zooplankton Feeding Efficiency		0.5-0.7			0.5-0.7			0.5-0.7	
NH_4 Decay Rate		0.16-0.20			0.16-0.20			0.16-0.20	
NO_2 Decay Rate		0.37-0.43			0.37-0.43			0.37-0.43	
Coliform Die-Off Rate		1.2-1.5			1.2-1.5			1.2-1.5	
Detritus Decay Rate		0.015-0.165			0.015-0.165			0.015-0.165	
Dissolved Organics Decay Rate		0.1-0.2			0.1-0.2			0.1-0.2	
Detritus Settling Rate		0.05-0.3			0.05-0.3			0.05-0.3	
ALGAE1 Update		0.1-0.64			0.1-0.64			0.1-0.64	
Alkalinity Update		207-287			170-284			208-294	
BOD Update		0.75-10.35			1.0-1.7			1.0-4.2	
NH_4 Update		0.01-0.12			0.01-0.195			0.01-0.08	
NO_3 Update		0.01-0.19			0.01-0.41			0.01-0.13	
Coliform Update		6-2000			6-3000			6-1000	
Detritus Update		1-6			1-6			1-6	
Phosphorus Update		0.011-0.047			0.01-0.05			0.004-0.04	

Table 7

Regression Equations and Generated Nutrient Loads to Proposed Twin Valley Lake

Equation	Phosphorus Load - $g/m^2/yr$		
	1971	1975	1976
$M = 0.0469\ F^{1.14}$	5.03	11.3	2.86
$C = 0.0432 + 0.0042F - 0.0000334F^2$	6.87	20.2	4.20
$\bar{C} = 0.057$	4.45	8.62	2.52
$\bar{C}_{75} = 0.079$	--	12.0	--
$\bar{C}_{76} = 0.056$	--	--	12.0
	Nitrogen Load - $g/m^2/yr$		
	1971	1975	1976
$M = 0.0622F^{1.26}$	8.84	22.9	5.10
$C = 0.0653 + 0.0143F - 0.000144F^2$	16.4	45.1	10.2
$\bar{C} = 0.11$	8.59	16.6	4.87
$\bar{C}_{75} = 0.23$	--	34.7	--
$\bar{C}_{76} = 0.074$	--	--	3.28

where M = mass, g

C = concentration, g/m^3

$\bar{C}$ = mean concentration, g/m^3

$\bar{C}_{75}$ = mean concentration in 1975, g/m^3

$\bar{C}_{76}$ = mean concentration in 1976, g/m^3

F = flow, m^3/sec

Table 8

Nutrient Loads to Proposed Twin Valley Impoundment

No.	Category	Year 1971	1975	1976
1	Critical Phosphorus Loads Based on In-Lake Concentrations of			
	L_c* (10 μg/ℓ) $g/m^2/yr$	0.96	1.76	0.58
	L_c (20 μg/ℓ) $g/m^2/yr$	1.93	3.53	1.17
	(Vollenweider 1976)			
2	Calculated Phosphorus Loads			
	Mass Regression Equation, $g/m^2/yr$	5.03	11.3	2.86
	Mean Conc. Mean Flow, $g/m^2/yr$	4.45	12.0	2.48
3	Critical Nitrogen Loads Based on N:P Ratios			
	L_c (10 μg/ℓ ρ**)			
	N:P-8:1 $g/m^2/yr$	7.70	14.1	4.60
	N:P-10:1 $g/m^2/yr$	9.60	17.6	5.80
	N:P-12:1 $g/m^2/yr$	11.5	21.1	7.00
	L_c (20 μg/ℓ ρ)			
	N:P-8:1 $g/m^2/yr$	15.4	28.2	9.40
	N:P-10:1 $g/m^2/yr$	19.3	35.3	11.7
	N:P-12:1 $g/m^2/yr$	23.2	42.4	14.0
4	Calculated Nitrogen Loads			
	Mass Regression Equation, $g/m^2/yr$	8.84	22.9	5.10
	Mean Conc. Mean Flow, $g/m^2/yr$	8.59	34.7	3.28

* L_c = critical annual phosphorus loading, $g/m^2/yr$.

** ρ = flushing rate per year, yr^{-1}.

Table 9

Classification, Discriminate, and Probability Functions For Predicting Oxic-Anoxic Lake Conditions (After Reckhow 1978)

Form	Equation
Classification Function, c.f.	$c.f. = 5.73 \log \bar{z} - 4.61 \log L + 4.10 \log \frac{\bar{z}}{\tau} - 11.50$
Classification Function, c.f.	$c.f. = 2.65 \log V - 2.50 \log m - 4.25$
Discriminant Function d.f.	$d.f. = \frac{117\ m^{1.63}}{V^{1.51}}$
Oxic Probability, P_{oxic}	$P_{oxic} = \frac{1}{e^{-(c.f.)} + 1}$
Anoxic Probability, P_{anoxic}	$P_{anoxic} = 1 - P_{oxic}$

where $\bar{z}$ = mean depth, m

L = annual areal P loadings, $g/m^2/yr$

τ = theoretical hydraulic residence time, yr

V = lake volume, $10^6\ m^3$

m = phosphorus mass, 10^3 kg P/yr

P = probability

Note: when $c.f. > 0 \Longrightarrow$ oxic conditions

$c.f. < 0 \Longrightarrow$ anoxic conditions

$d.f. < 1 \Longrightarrow$ oxic conditions

$d.f. > 1 \Longrightarrow$ anoxic conditions

Table 10

Probability of Anoxic Conditions Computed Using Classification Function vs. Discrimination Function (Reckhow 1978)

Equation	1971 Mass	1971 Conc	1975 Mass	1975 Conc	1976 Mass	1976 Conc
P_{anoxic} (c.f.)	0.97	0.96	0.98	0.96	0.96	0.95
P_{anoxic} (d.f.)	0.99	0.98	0.99	0.99	0.98	0.97

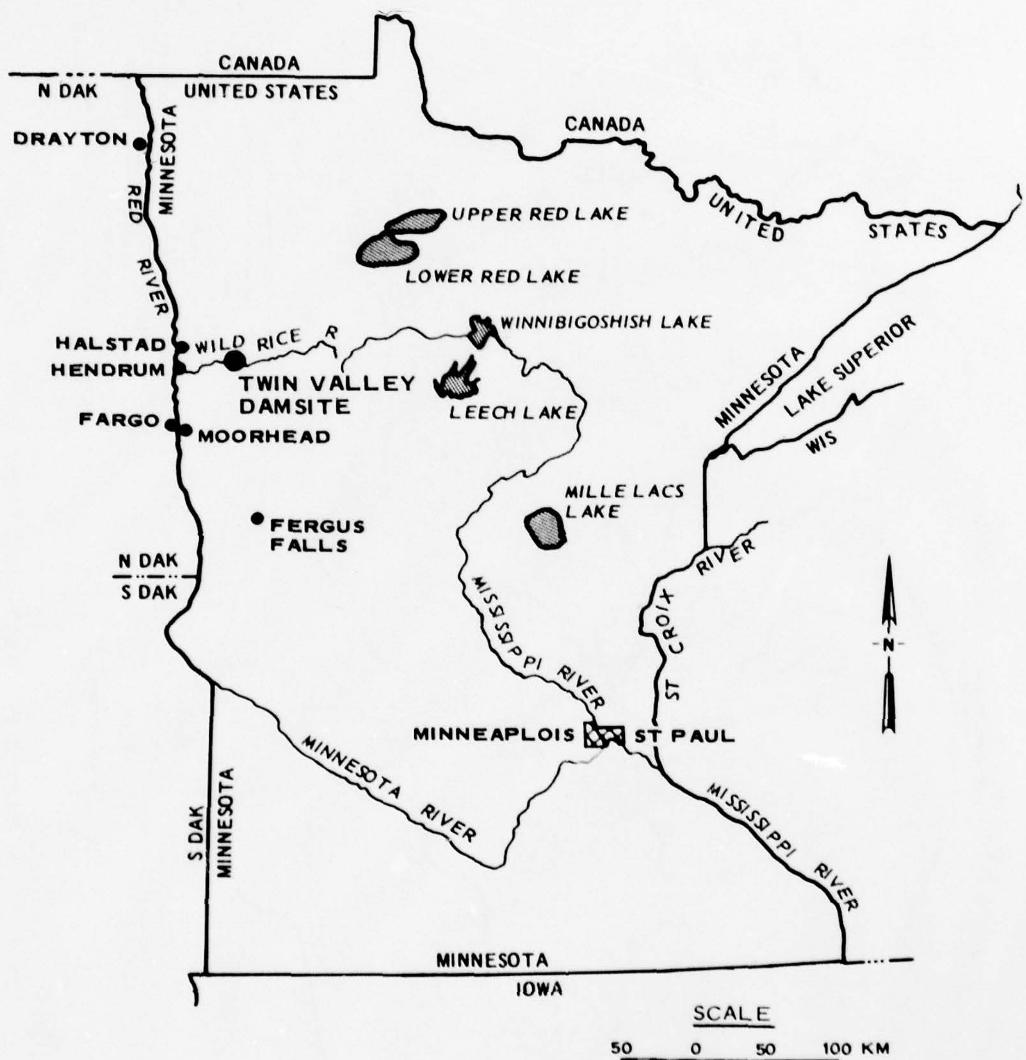

Figure 1. Location of project site

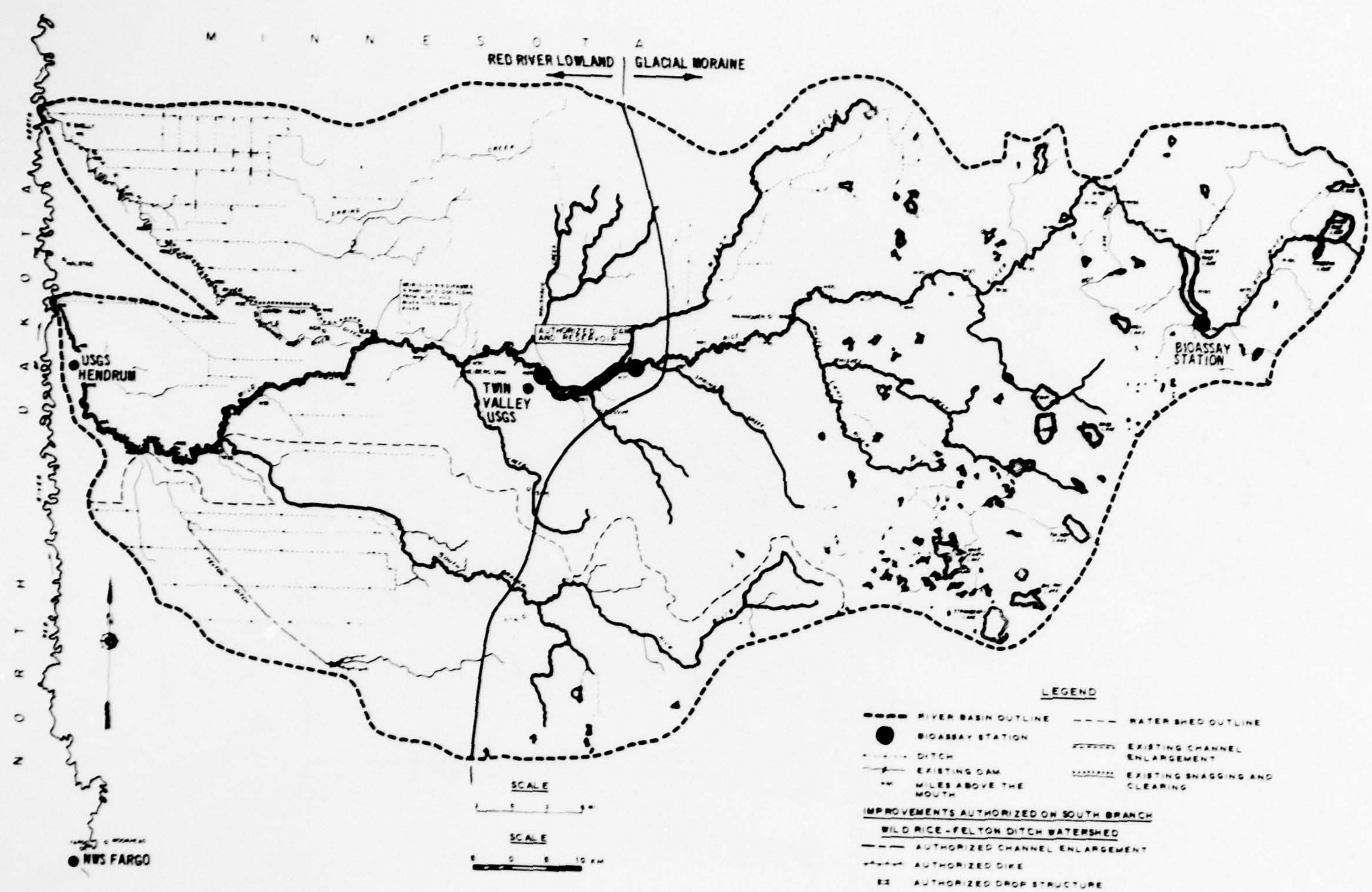

Figure 2. Wild Rice River Basin

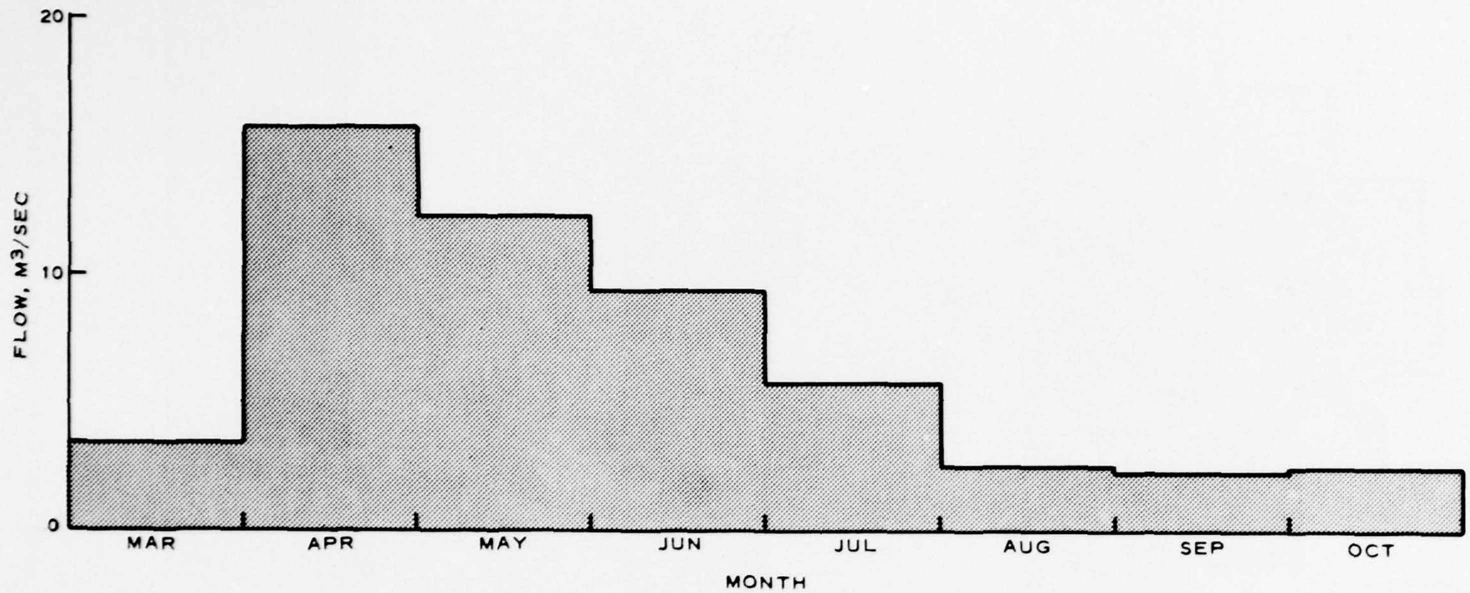

Figure 3. Mean monthly streamflow in the Wild Rice River at Twin Valley, Minn.

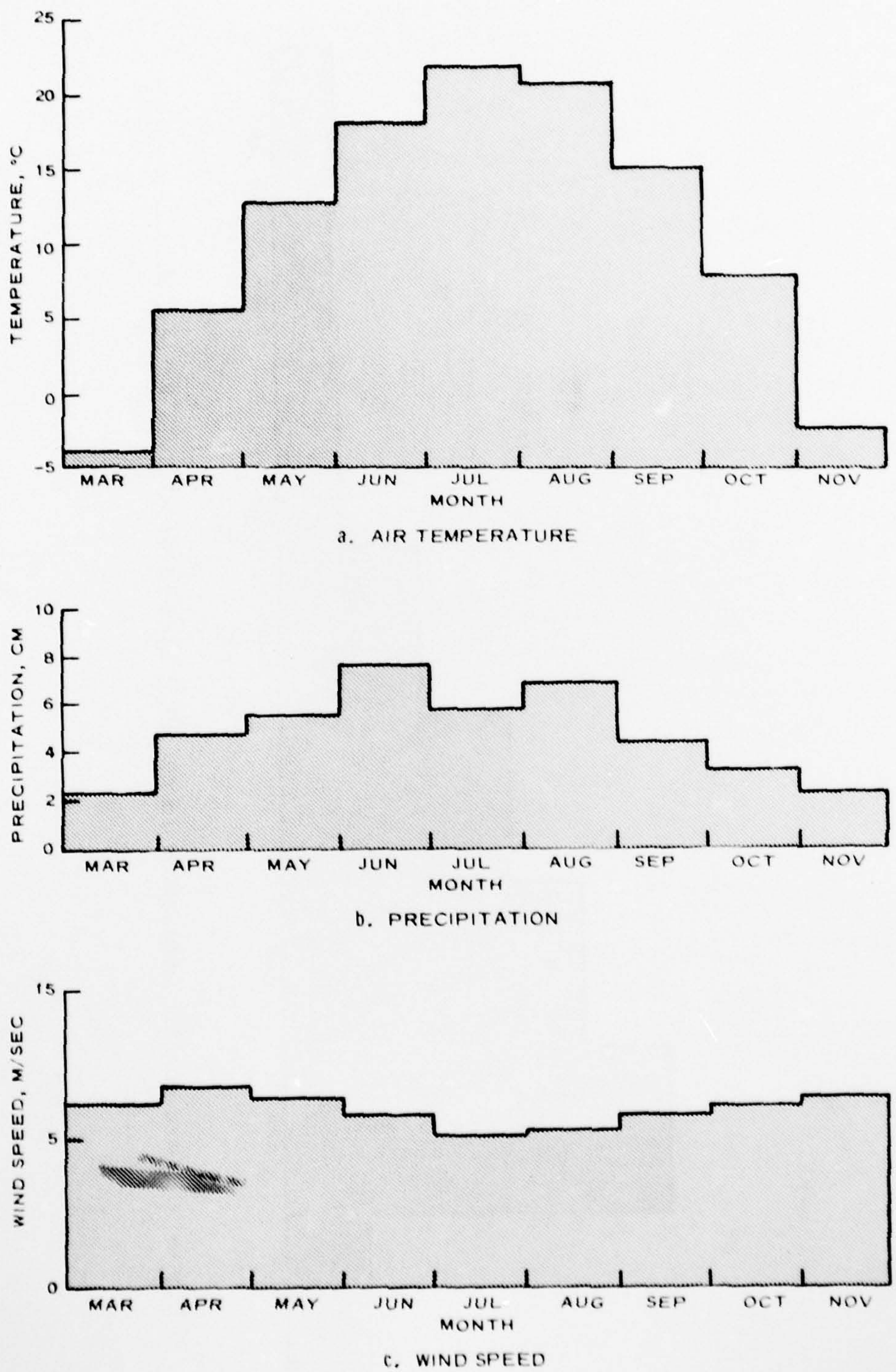

Figure 4. Mean monthly meteorological conditions at Fargo, N. D.

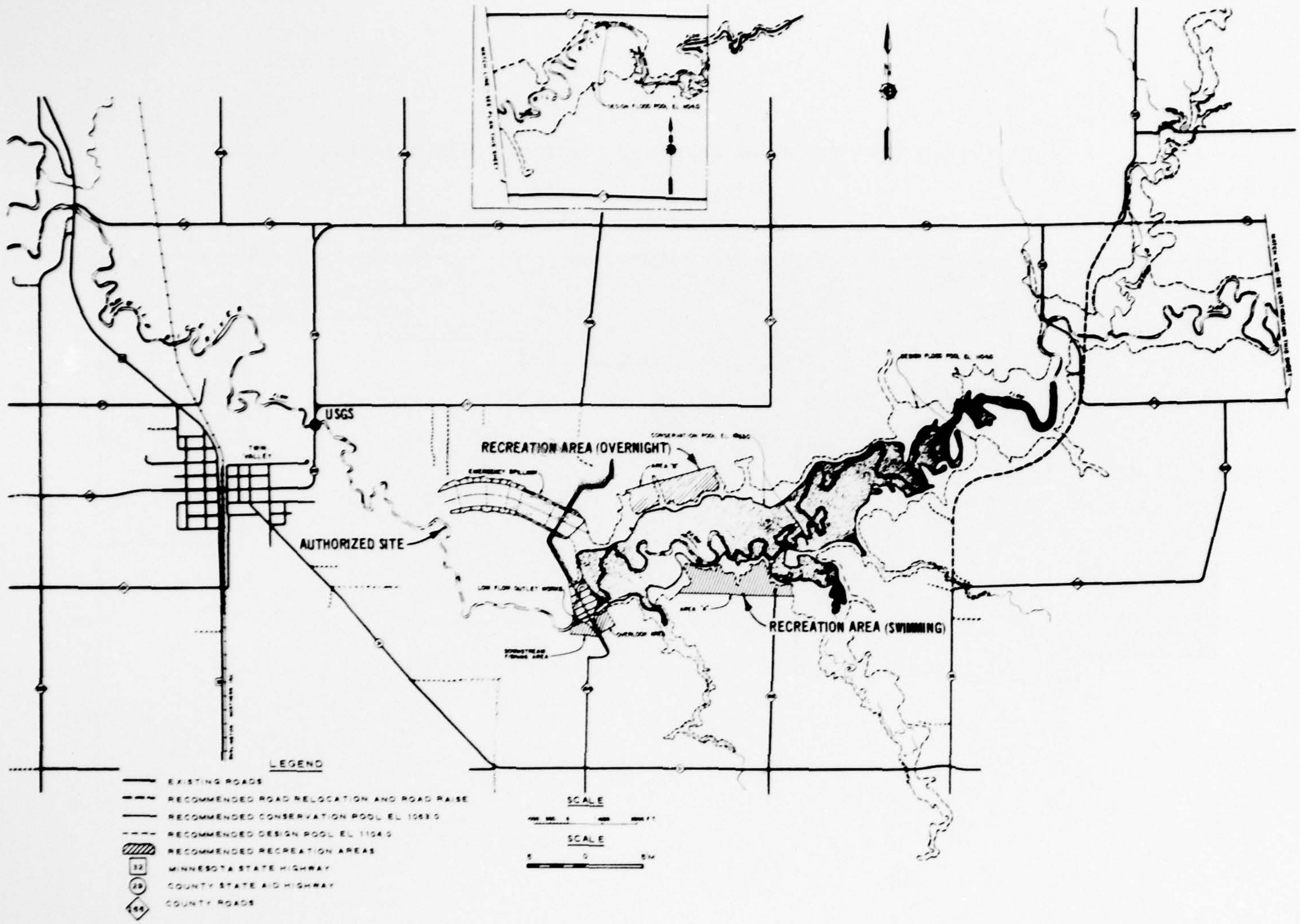

Figure 5. Proposed Twin Valley Reservoir (alternative site)

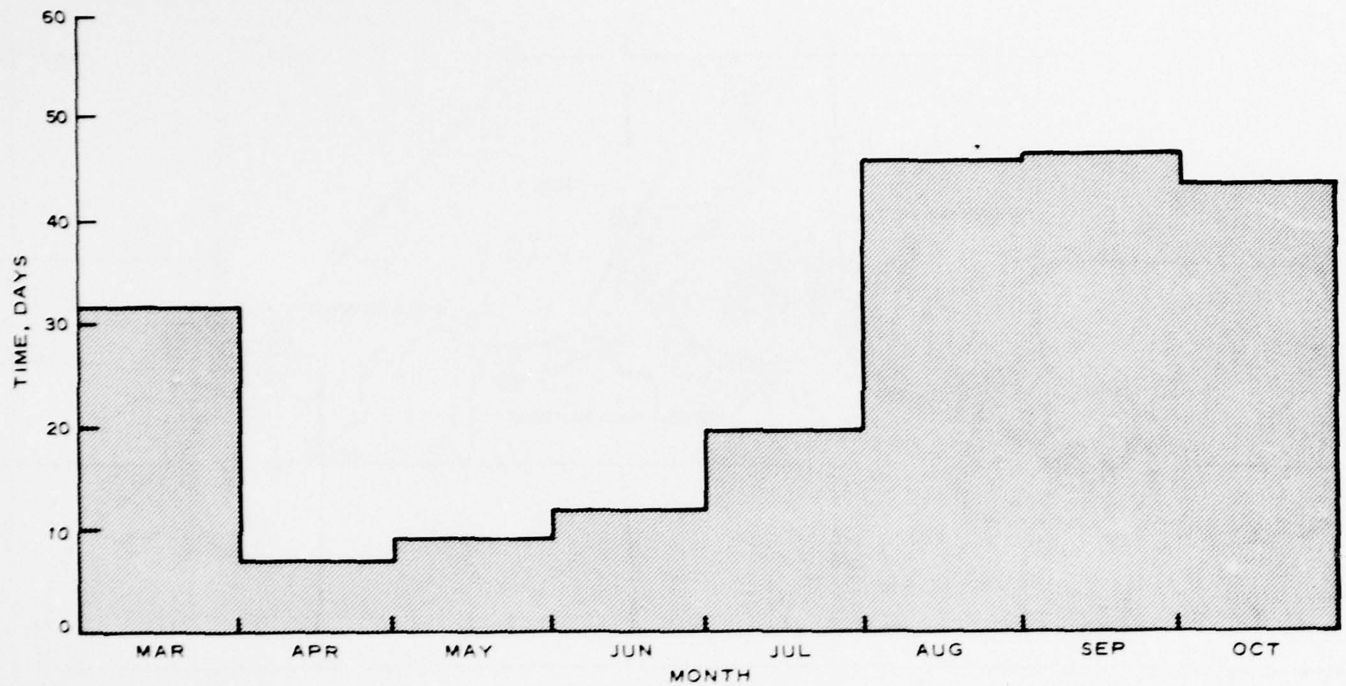

Figure 6. Mean monthly residence time for proposed Twin Valley Lake

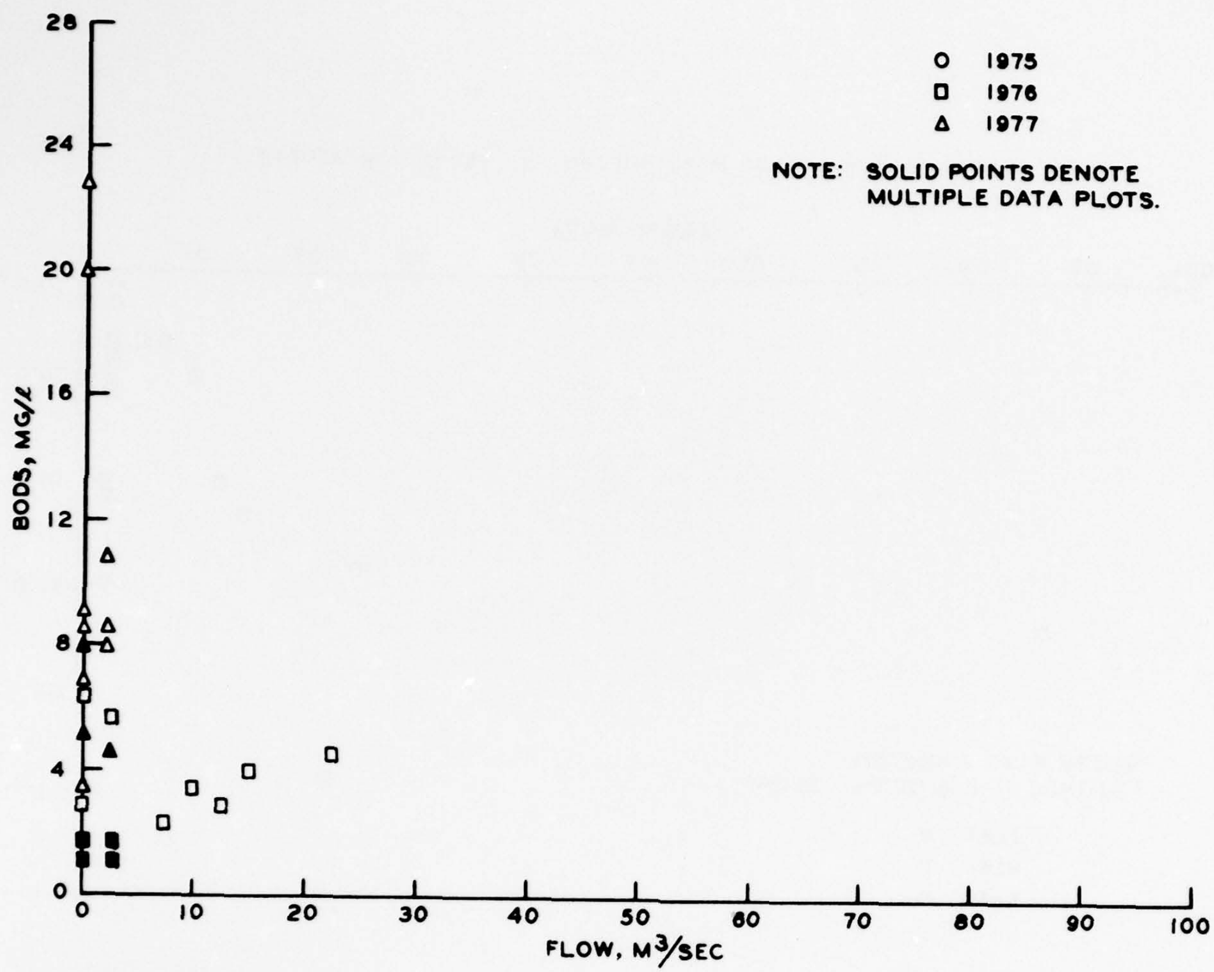

Figure 7. BOD5 versus flow at the damsite

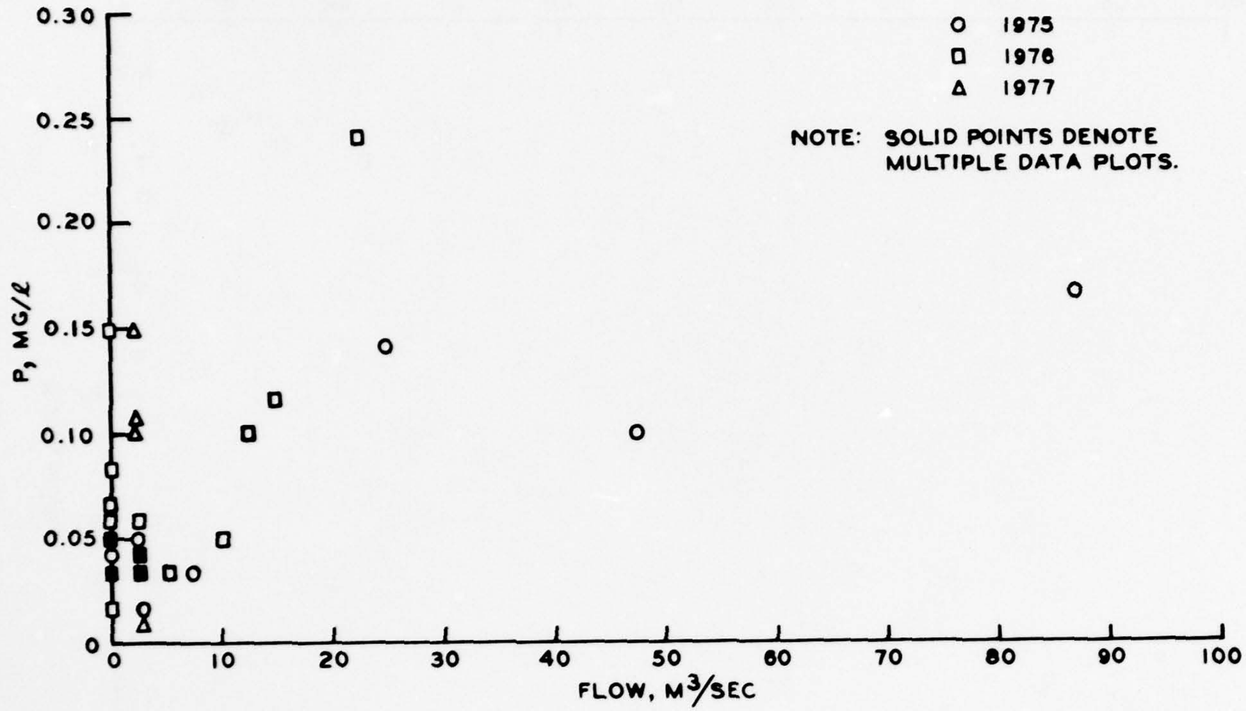

Figure 8. Total P versus flow at the damsite

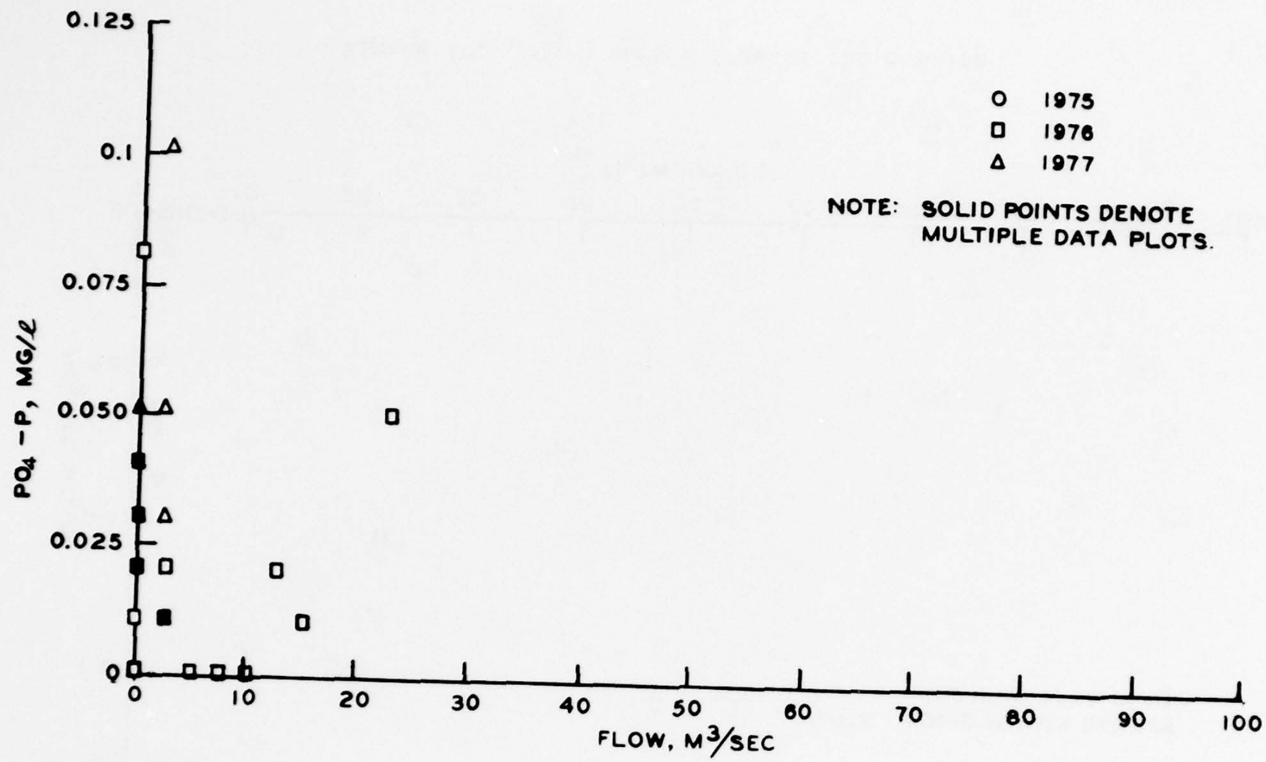

Figure 9. PO_4-P versus flow at the damsite

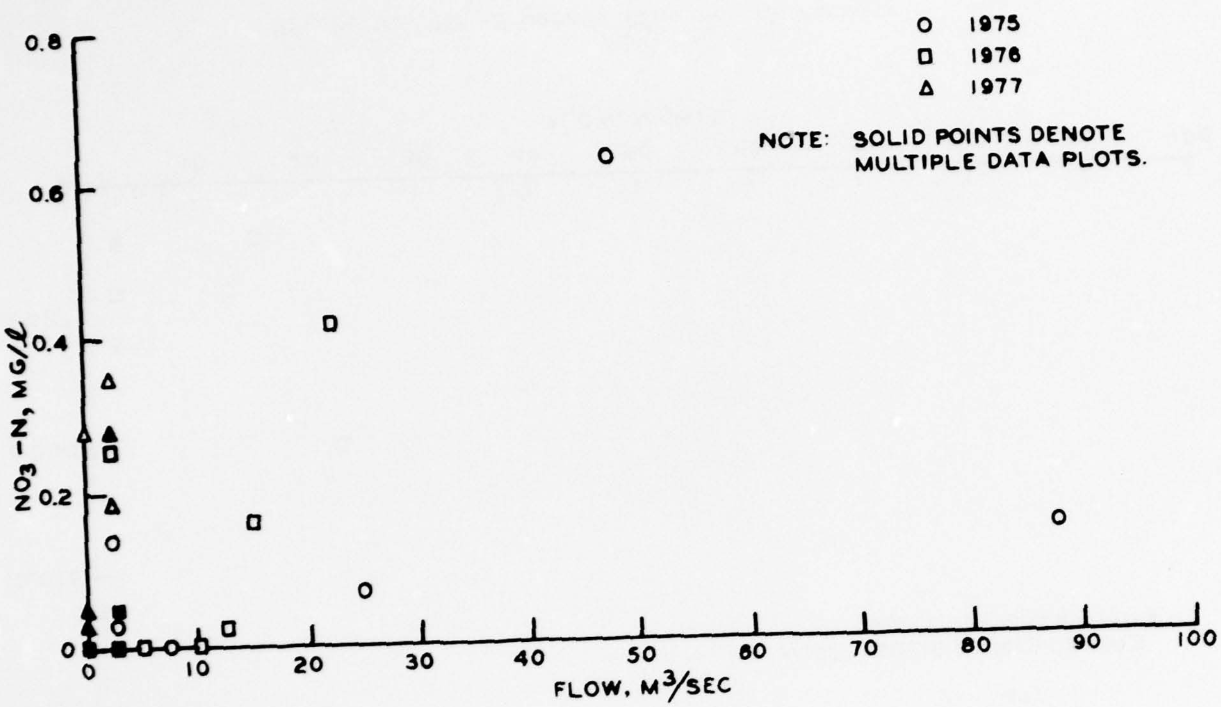

Figure 10. NO_3-N versus flow at the damsite

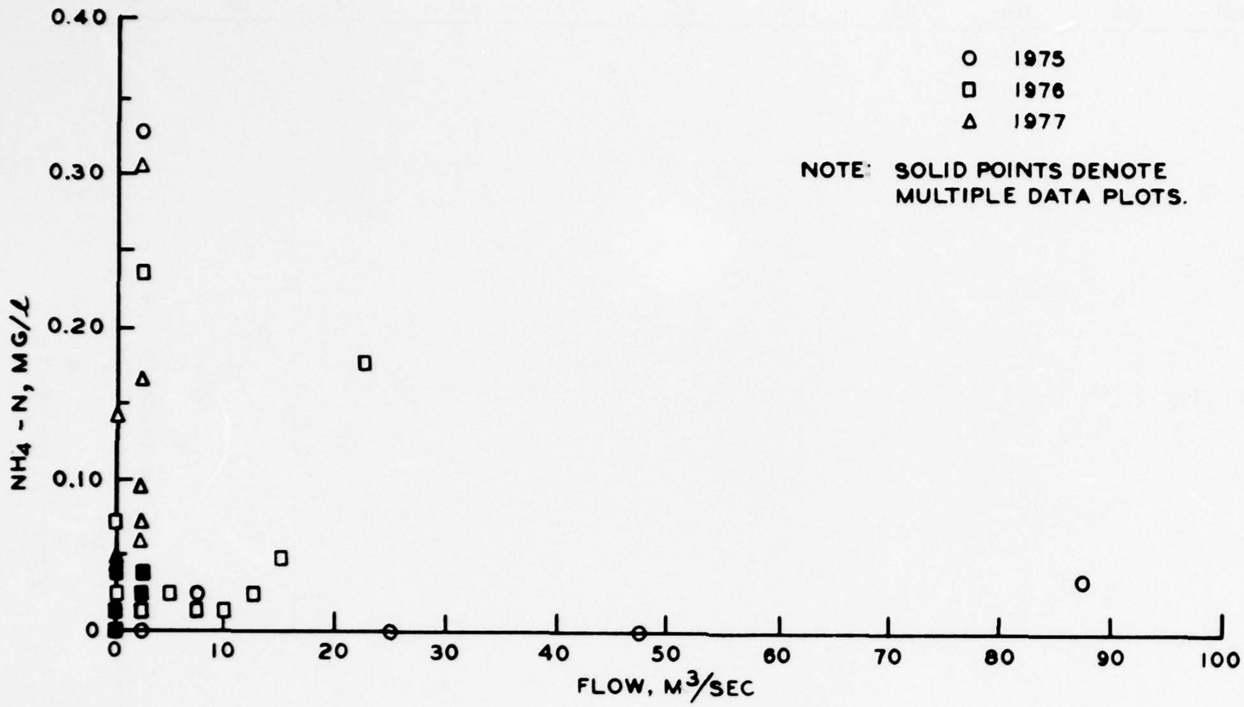

Figure 11. NH_4-N versus flow at the damsite

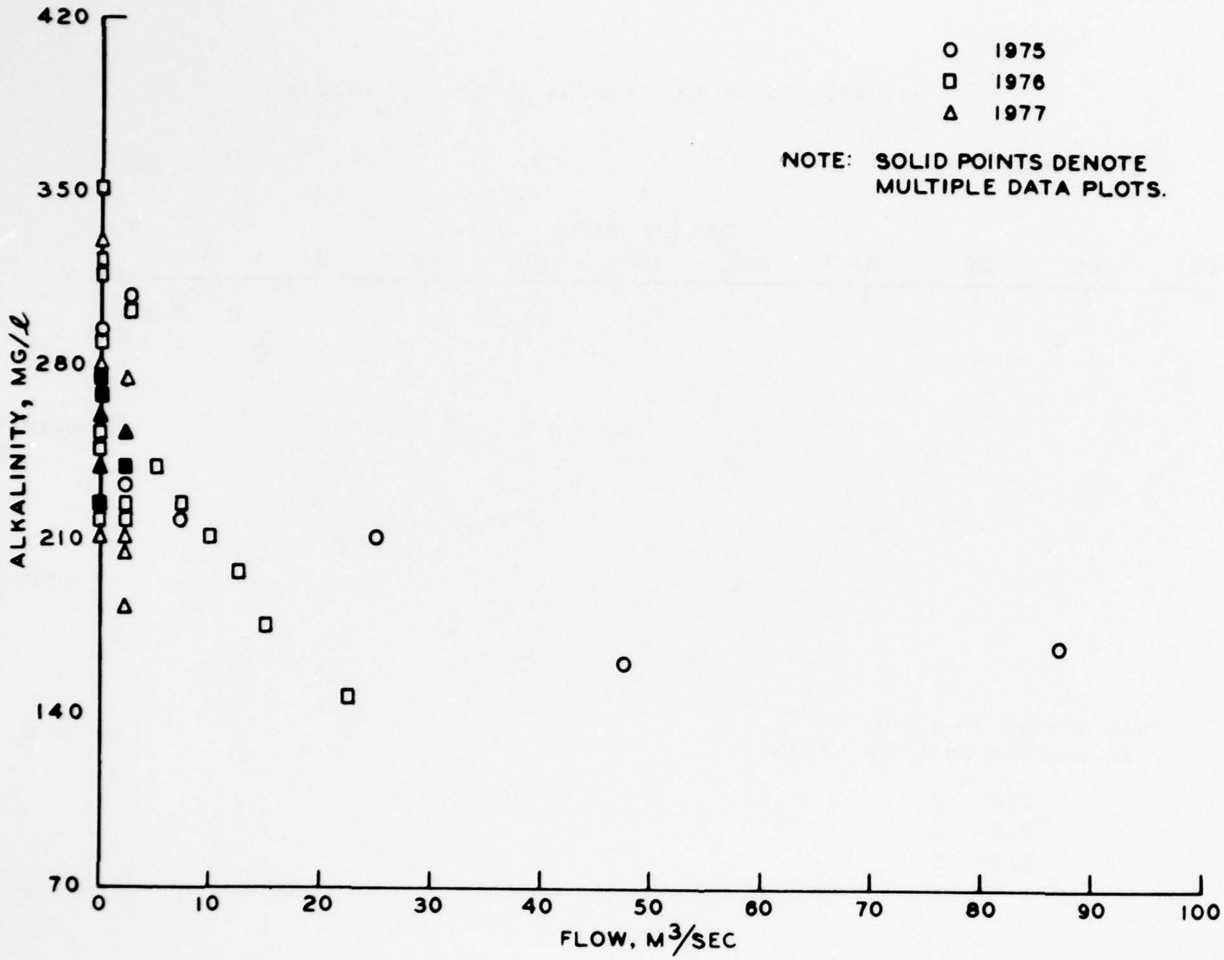

Figure 12. Alkalinity versus flow at the damsite

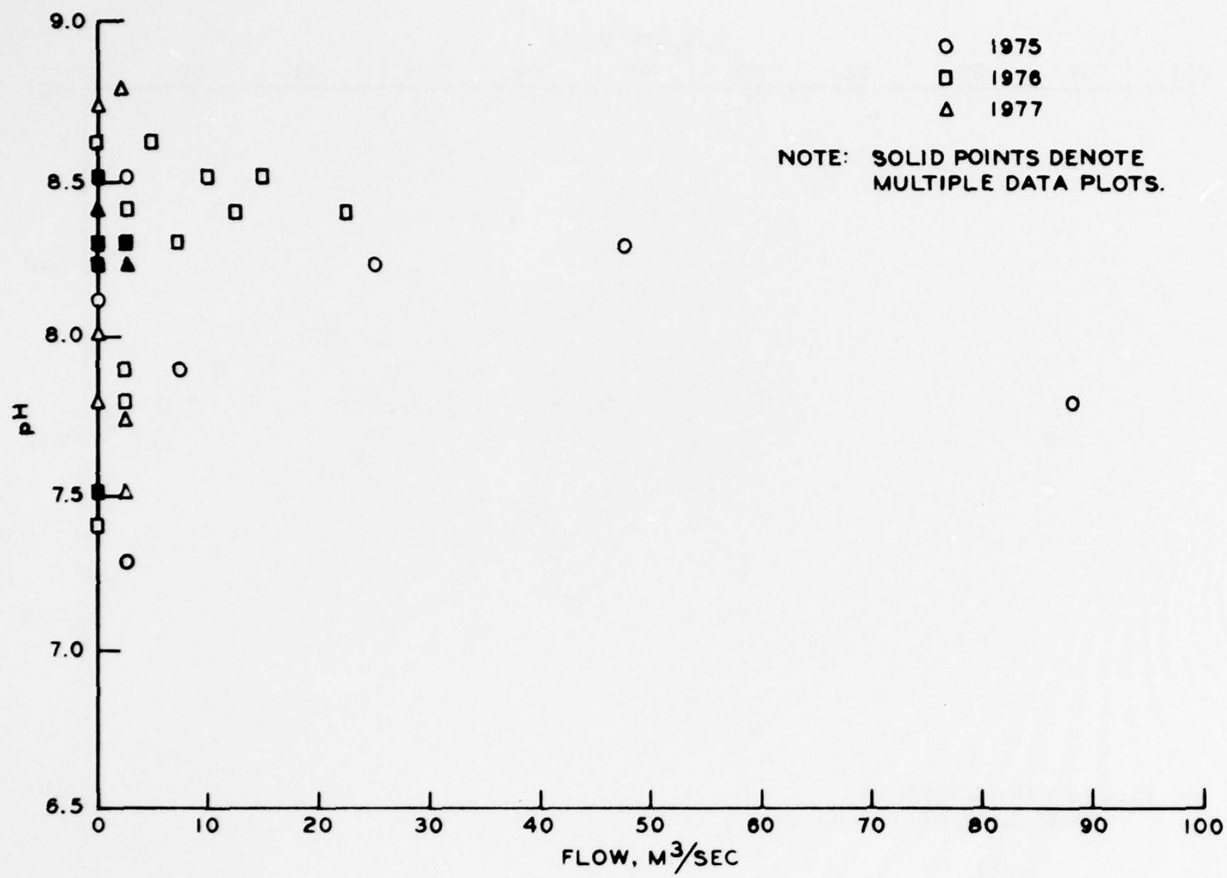

Figure 13. pH versus flow at the damsite

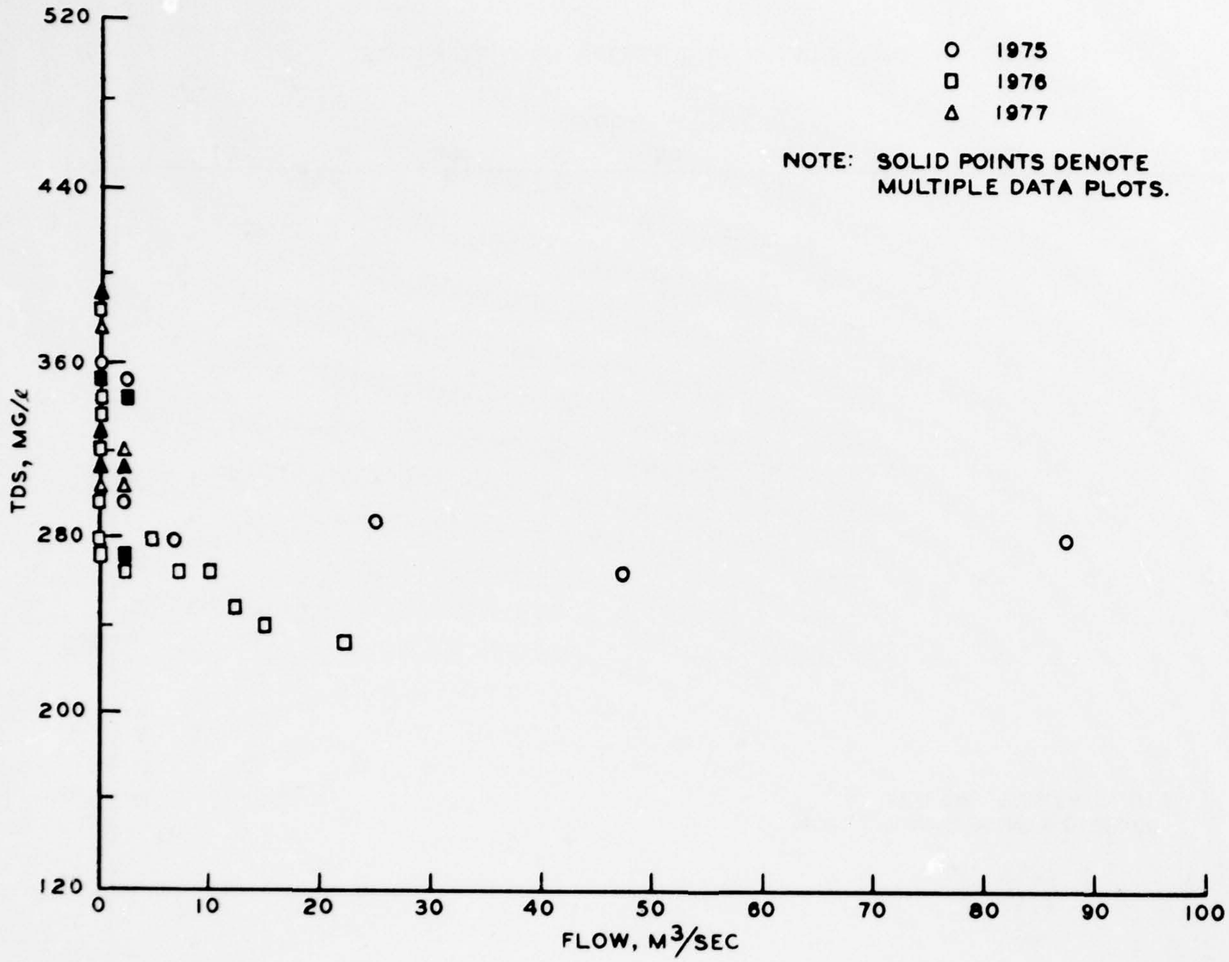

Figure 14. TDS versus flow at the damsite

AD-A074 550 ARMY ENGINEER WATERWAYS EXPERIMENT STATION VICKSBURG--ETC F/G 13/2
WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE, WILD RIC--ETC(U)
JUL 79 D E FORD, K W THORNTON, W B FORD

UNCLASSIFIED WES-EL-79-5 NL

2 OF 4

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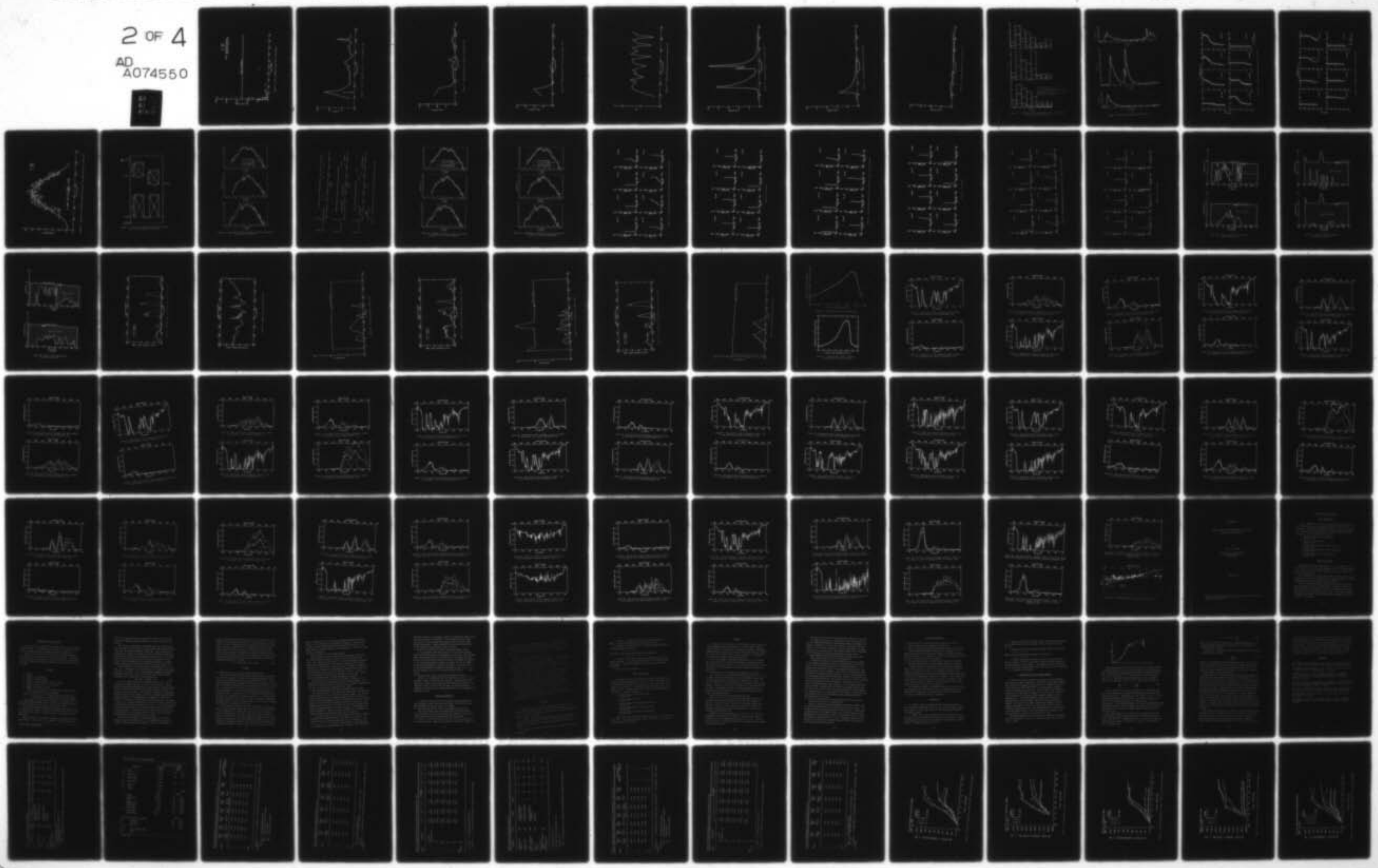
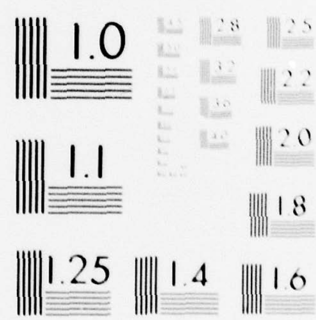
A074550





MICROCOPY RESOLUTION TEST CHART
NATIONAL BUREAU OF STANDARDS 1963-A

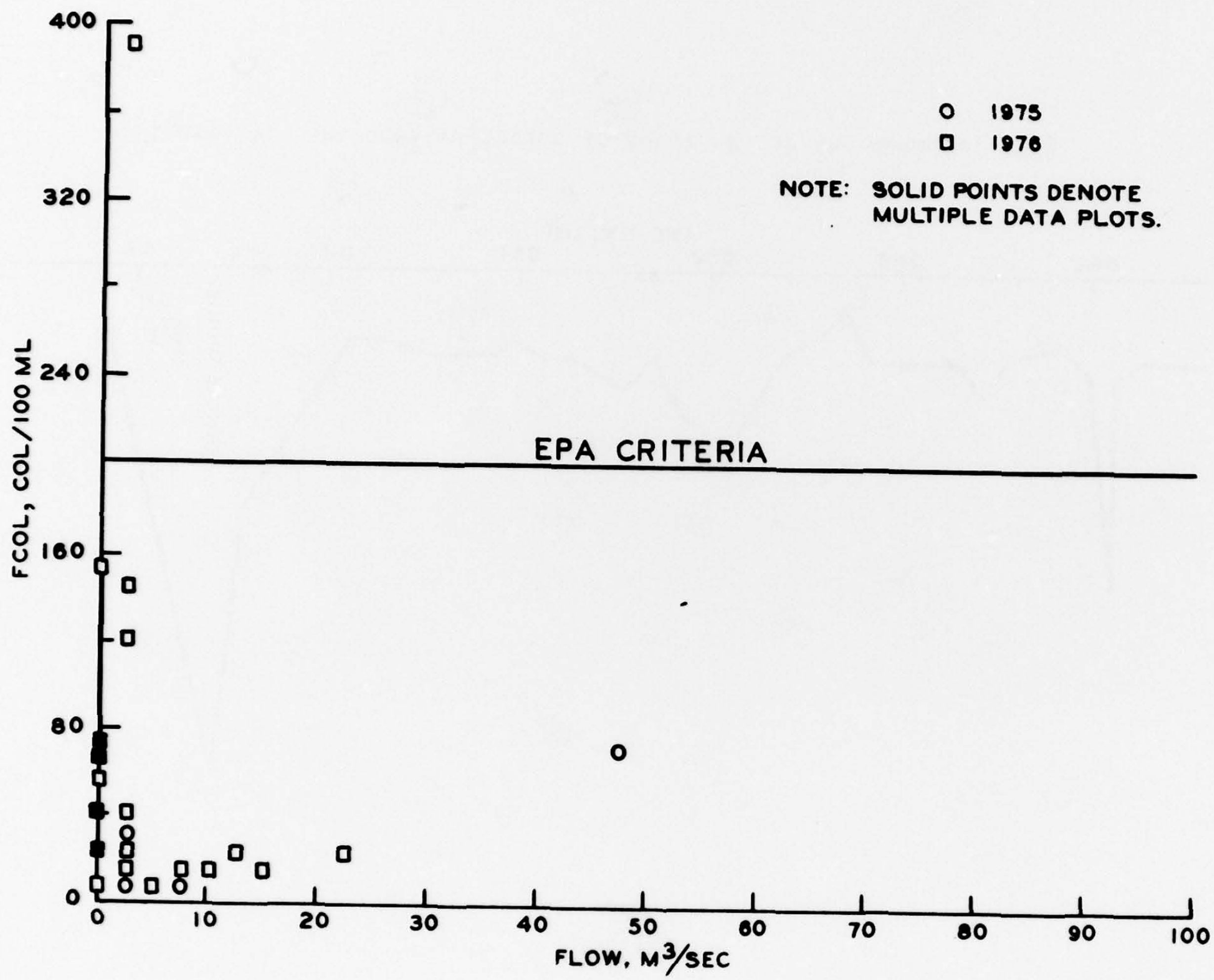


Figure 15. Fecal coliforms versus flow at the damsite

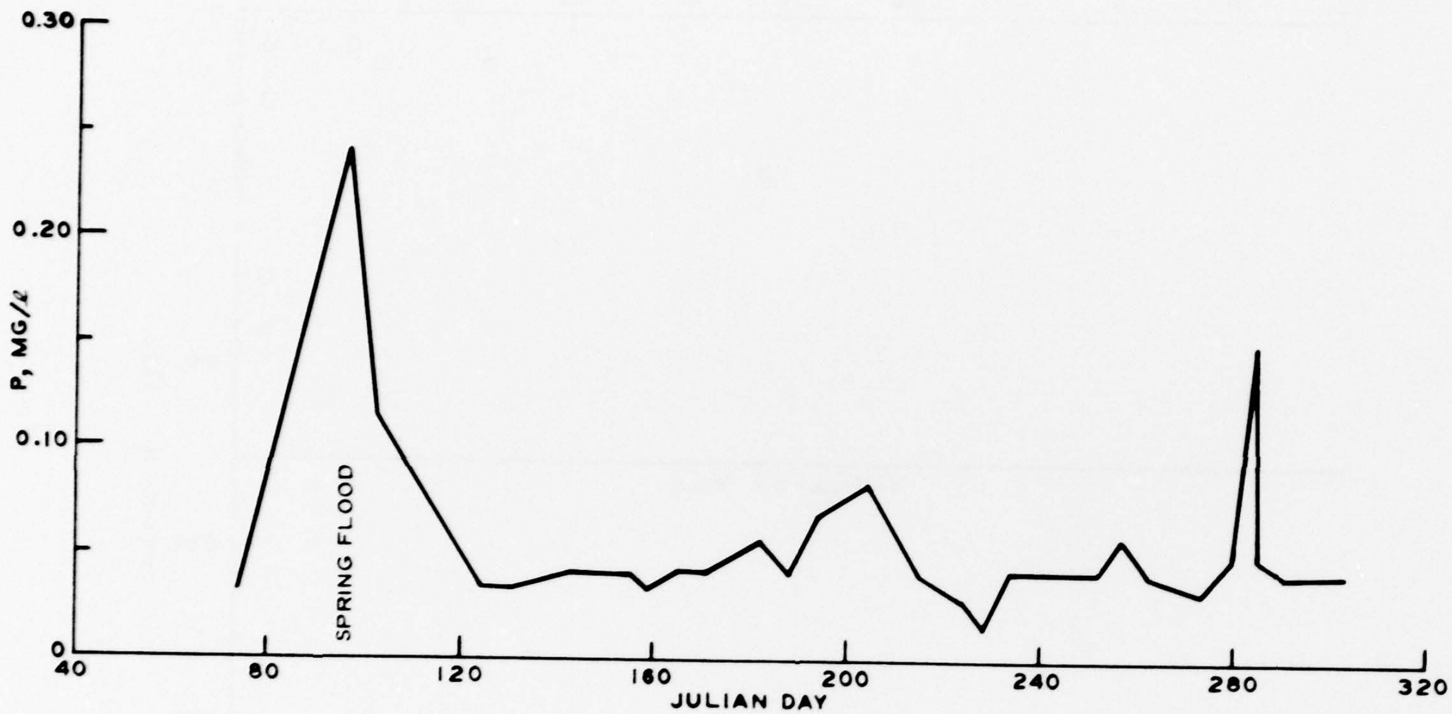


Figure 16. Seasonal variation in total P at the damsite, 1976

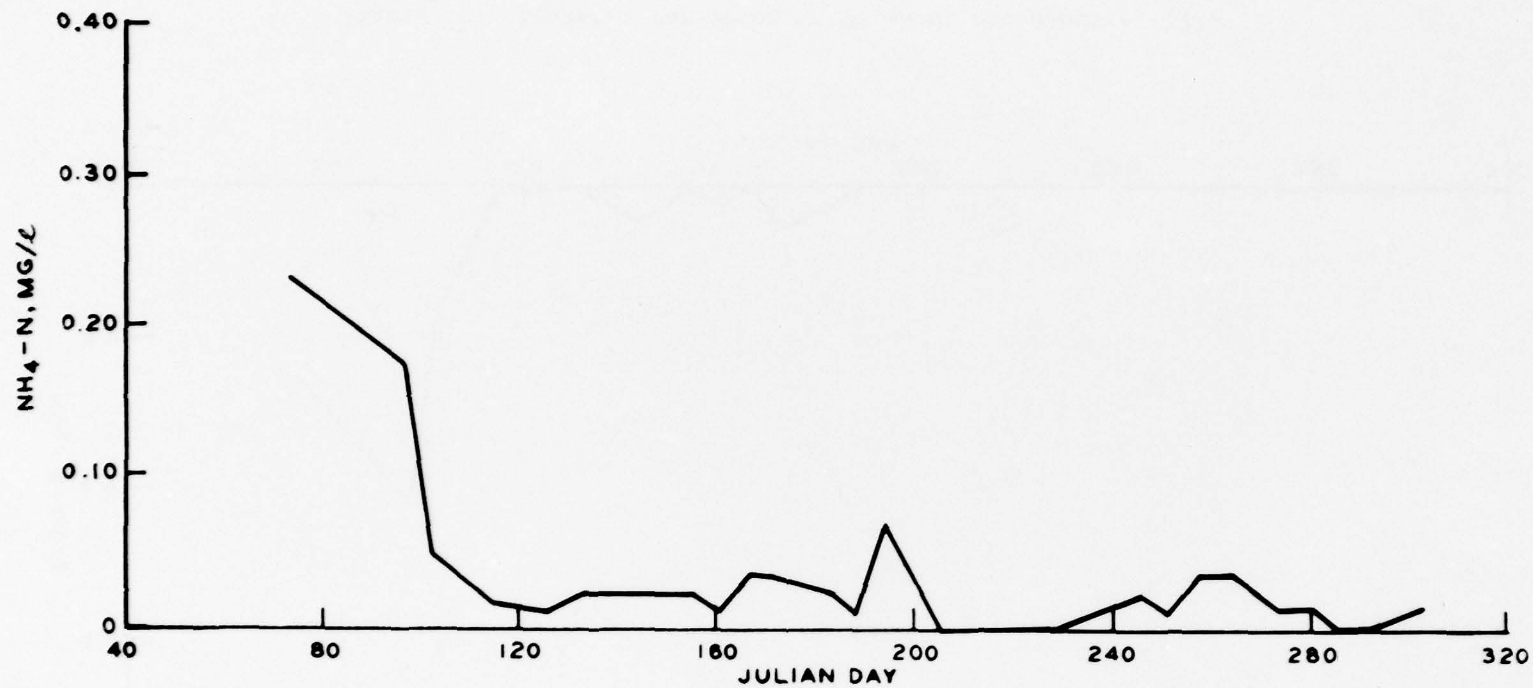


Figure 17. Seasonal variation in NH_4-N at the damsite, 1976

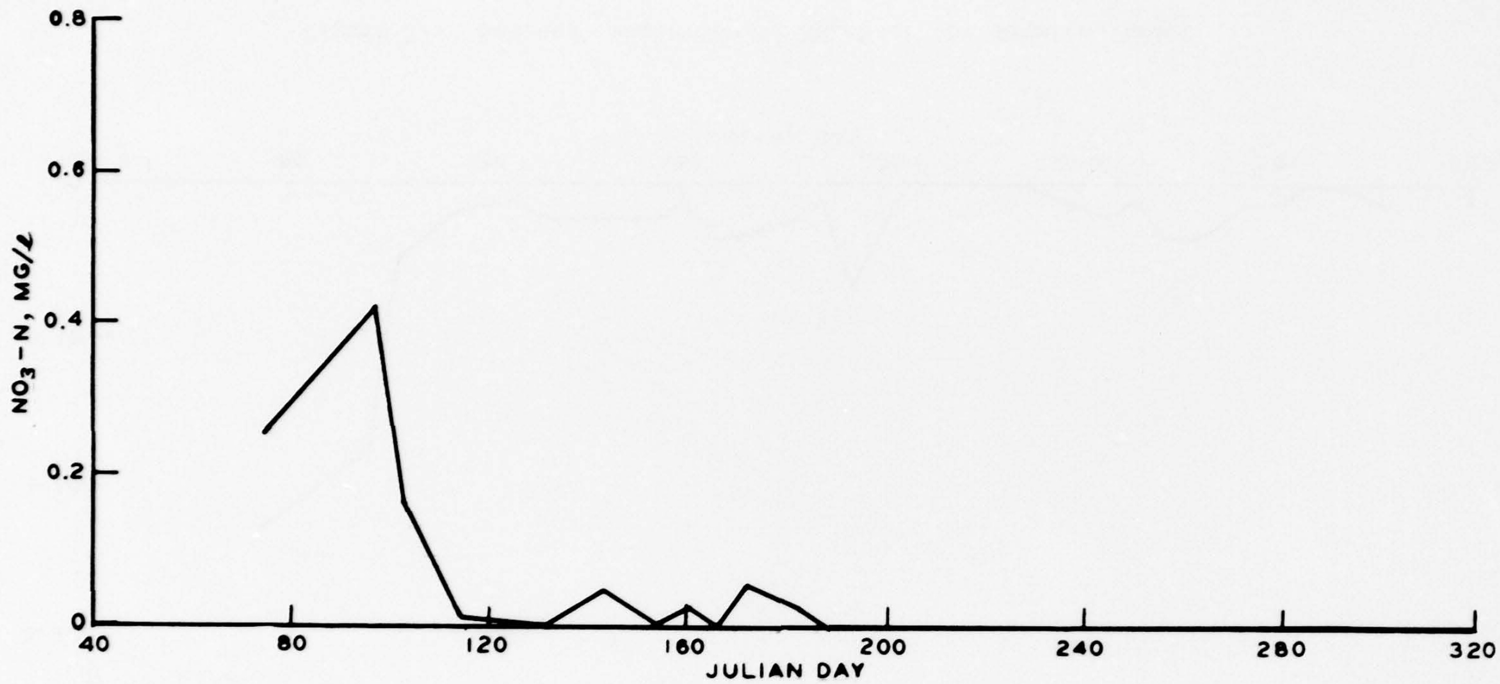


Figure 18. Seasonal variation in NO_3-N at the damsite, 1976

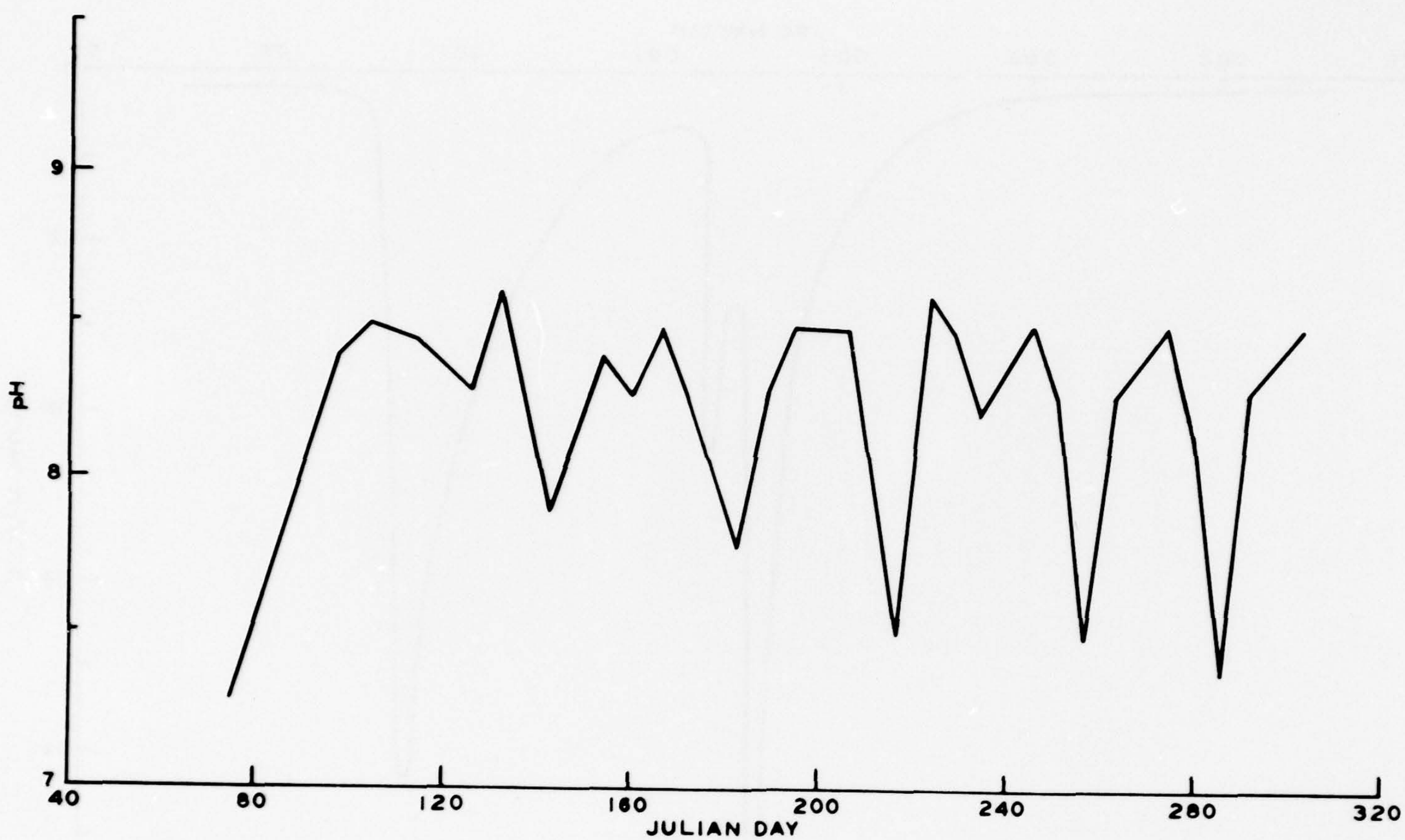


Figure 19. Seasonal variation in pH at the damsite, 1976

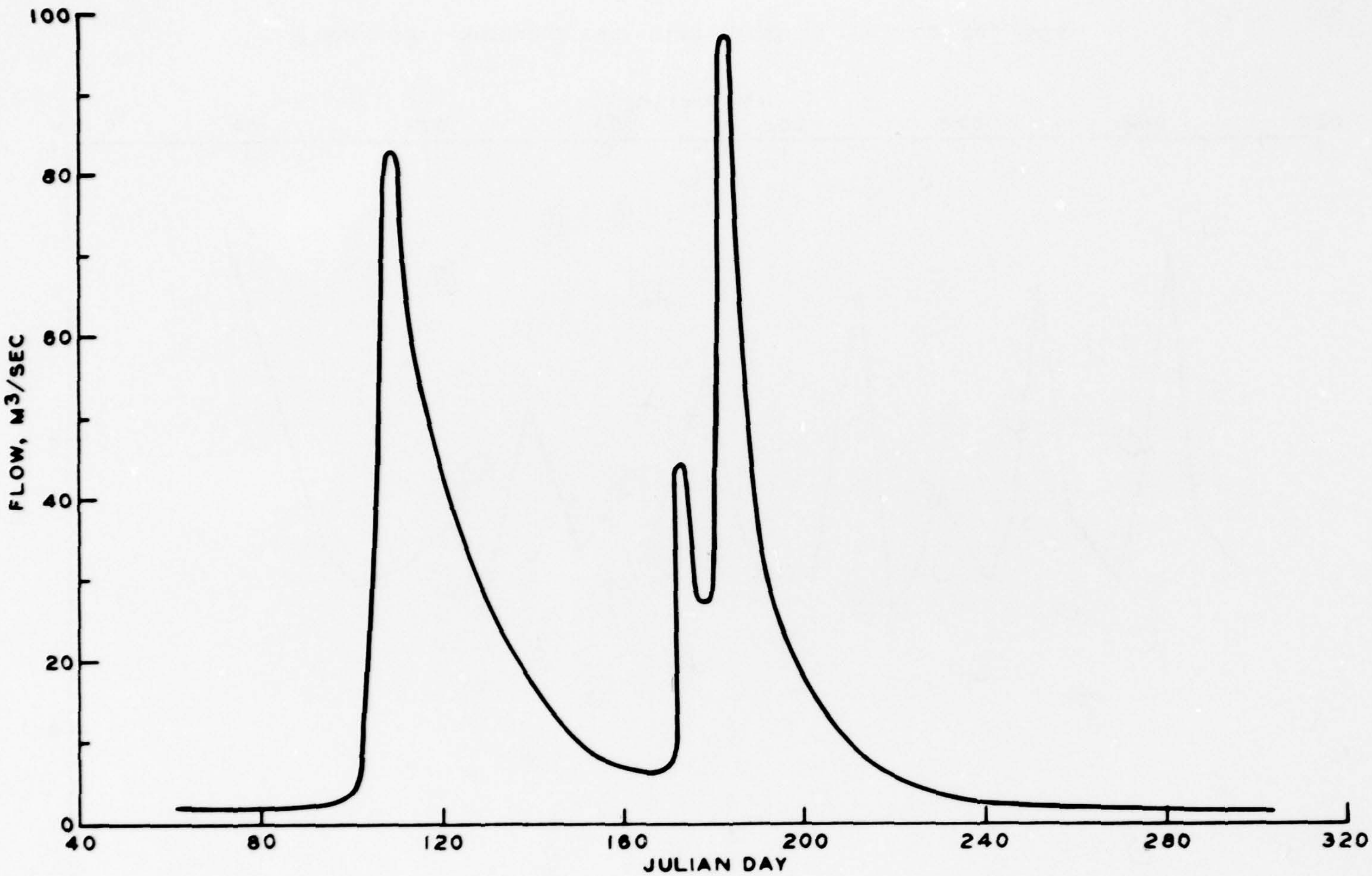


Figure 20. Seasonal hydrograph at the damsite, 1975

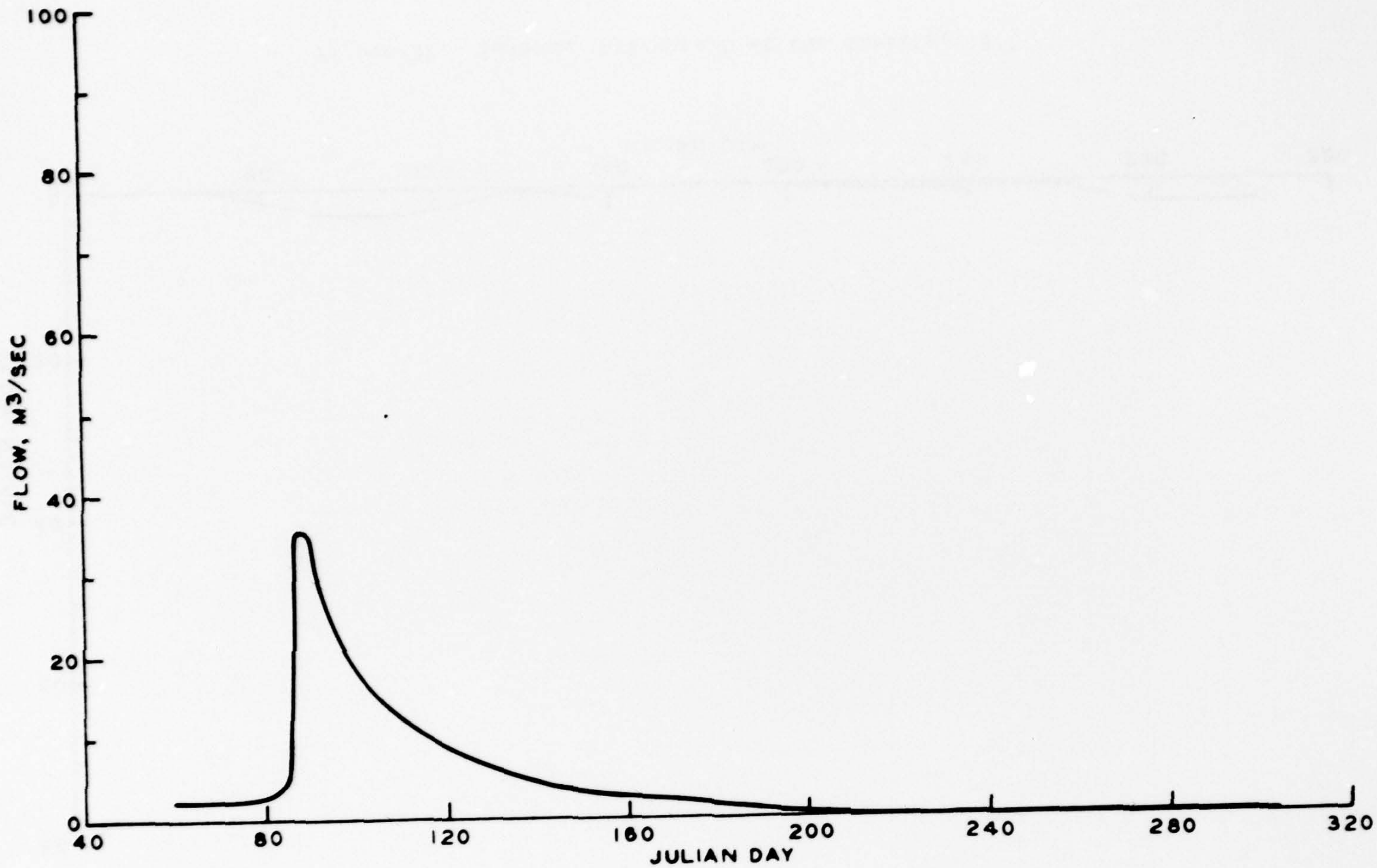


Figure 21. Seasonal hydrograph at the damsite, 1976

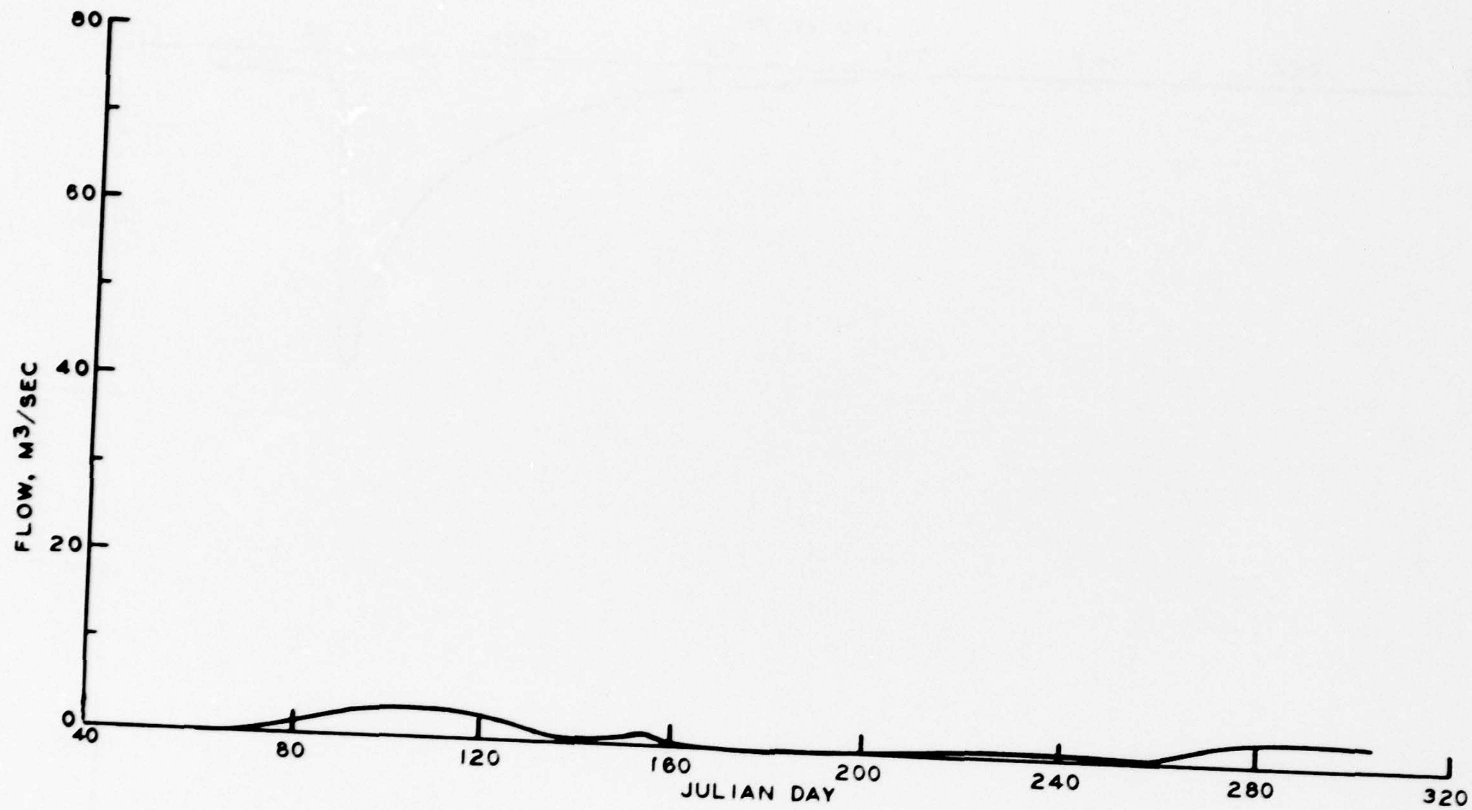


Figure 22. Seasonal hydrograph at the damsite, 1977

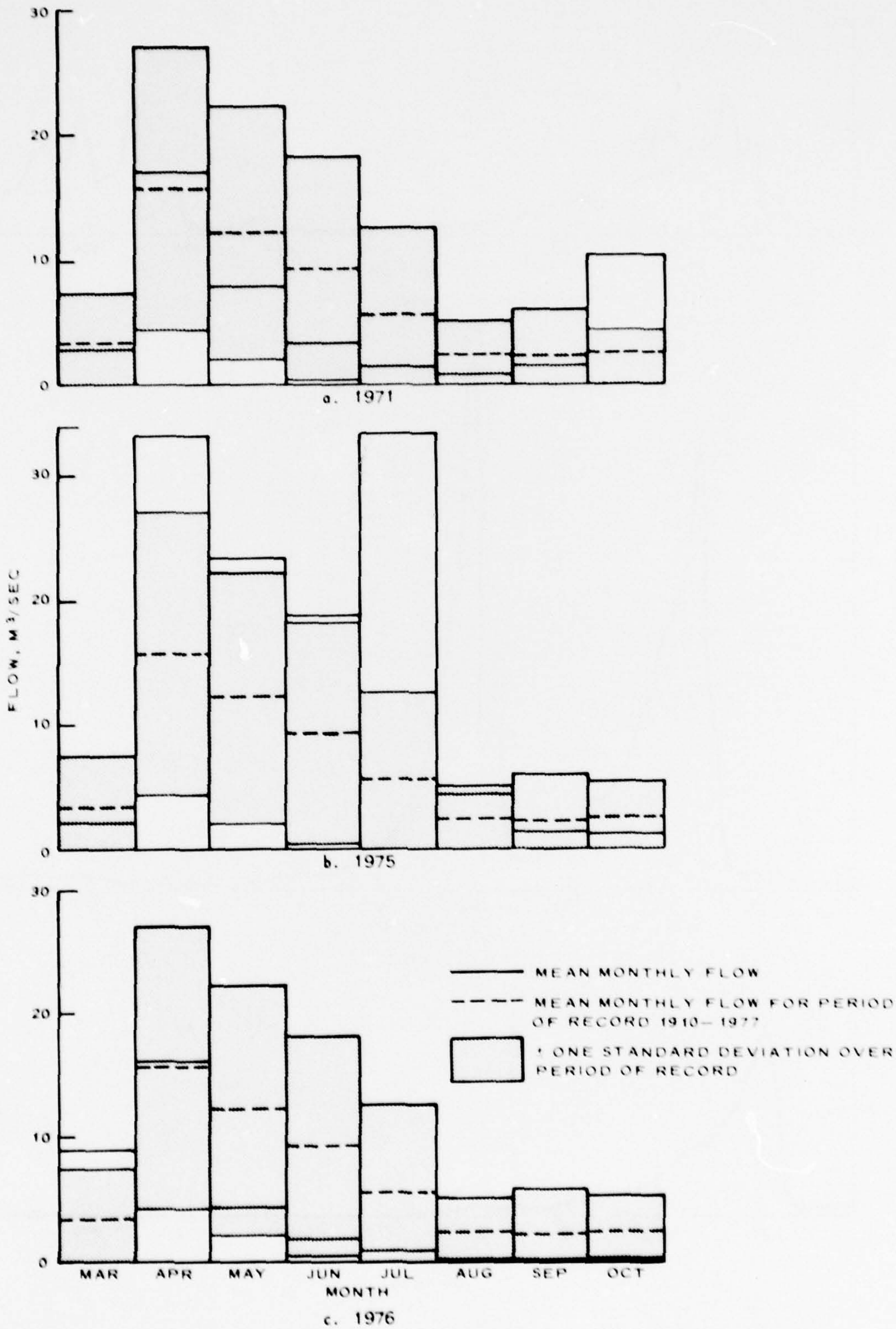


Figure 23. Comparison of the mean monthly flows for the period of record with the study years

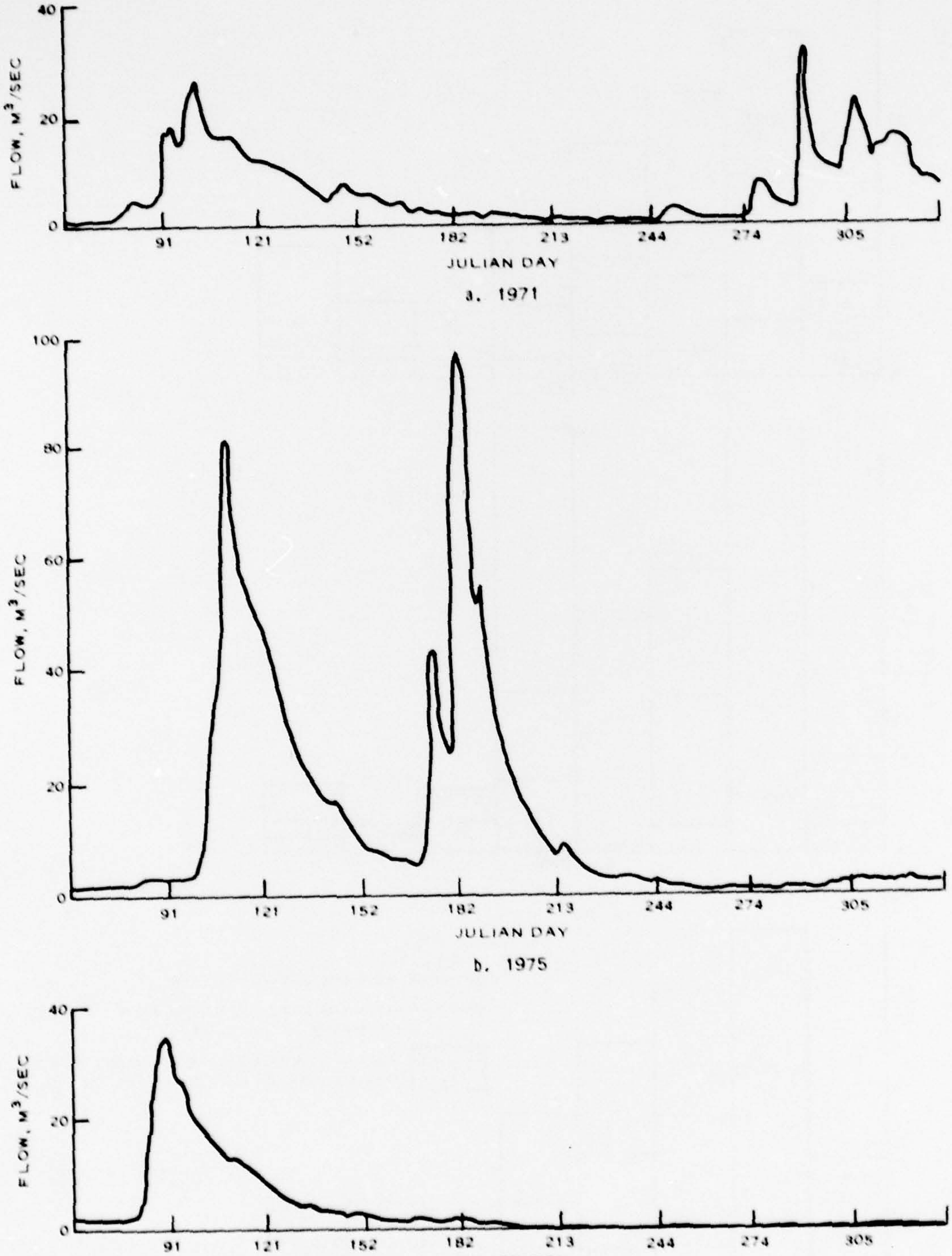


Figure 24. Mean daily discharge at Twin Valley

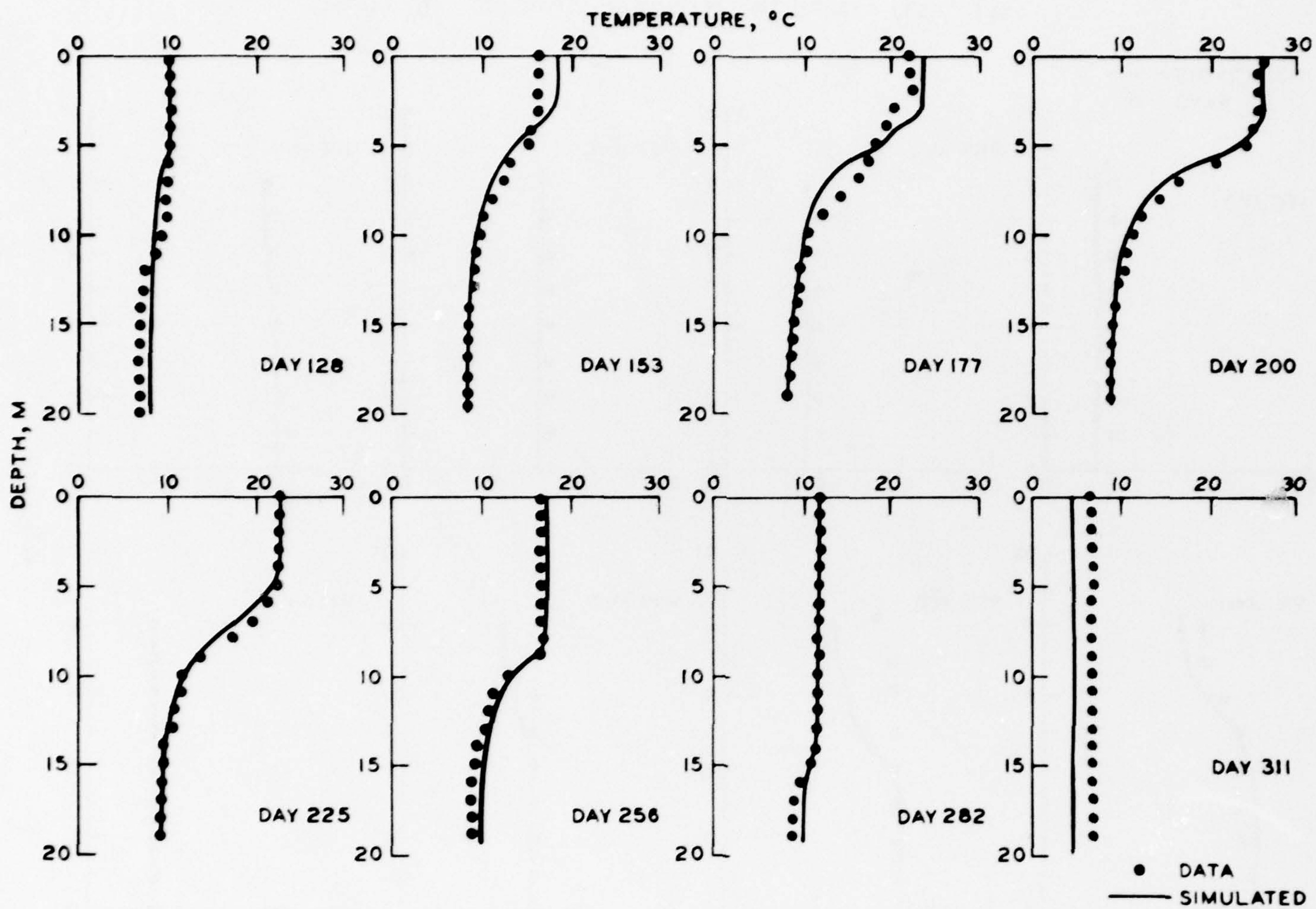


Figure 25. Calibration simulation, Lake Calhoun, 1974

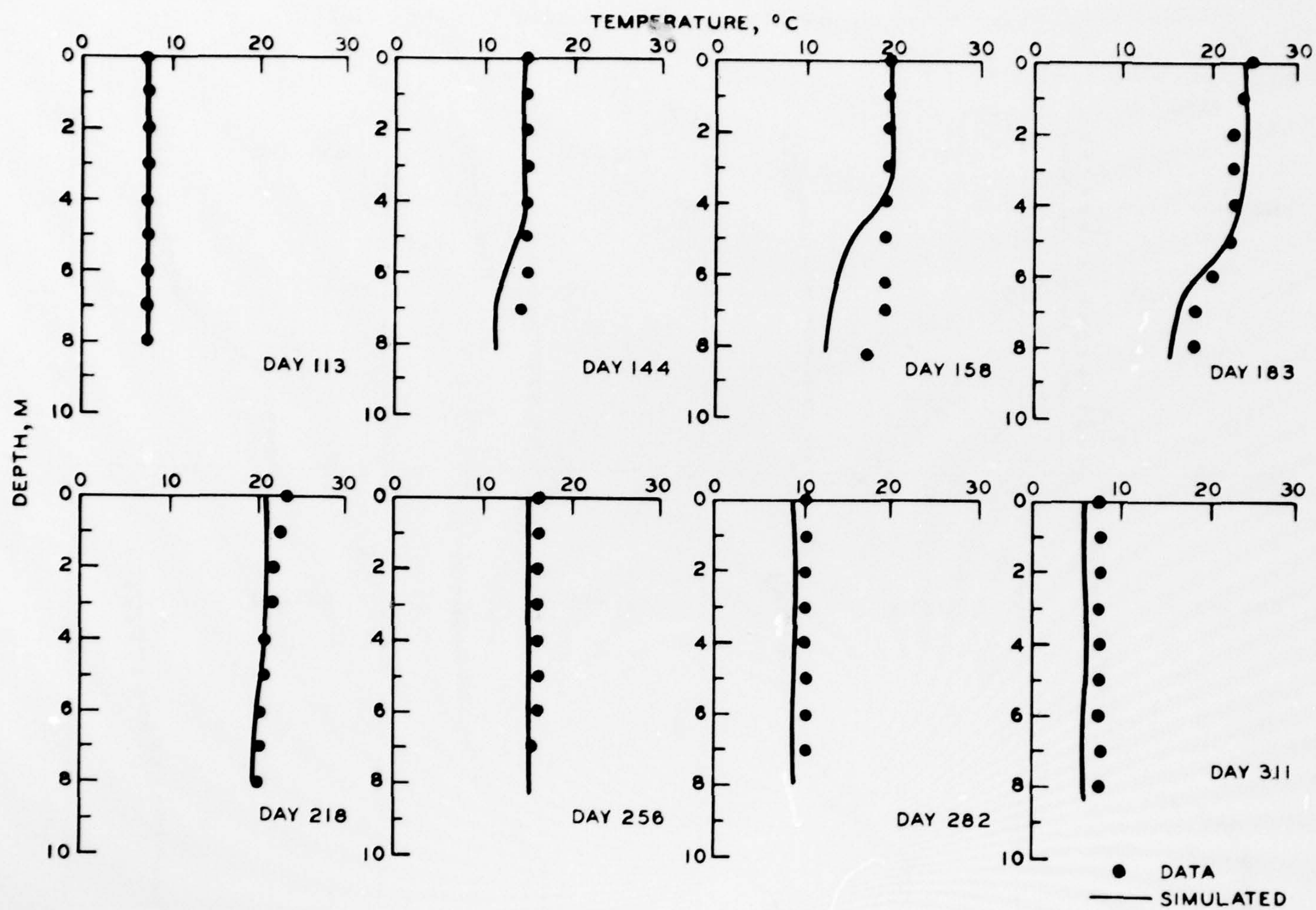


Figure 26. Calibration simulation, Turtle Lake, 1974

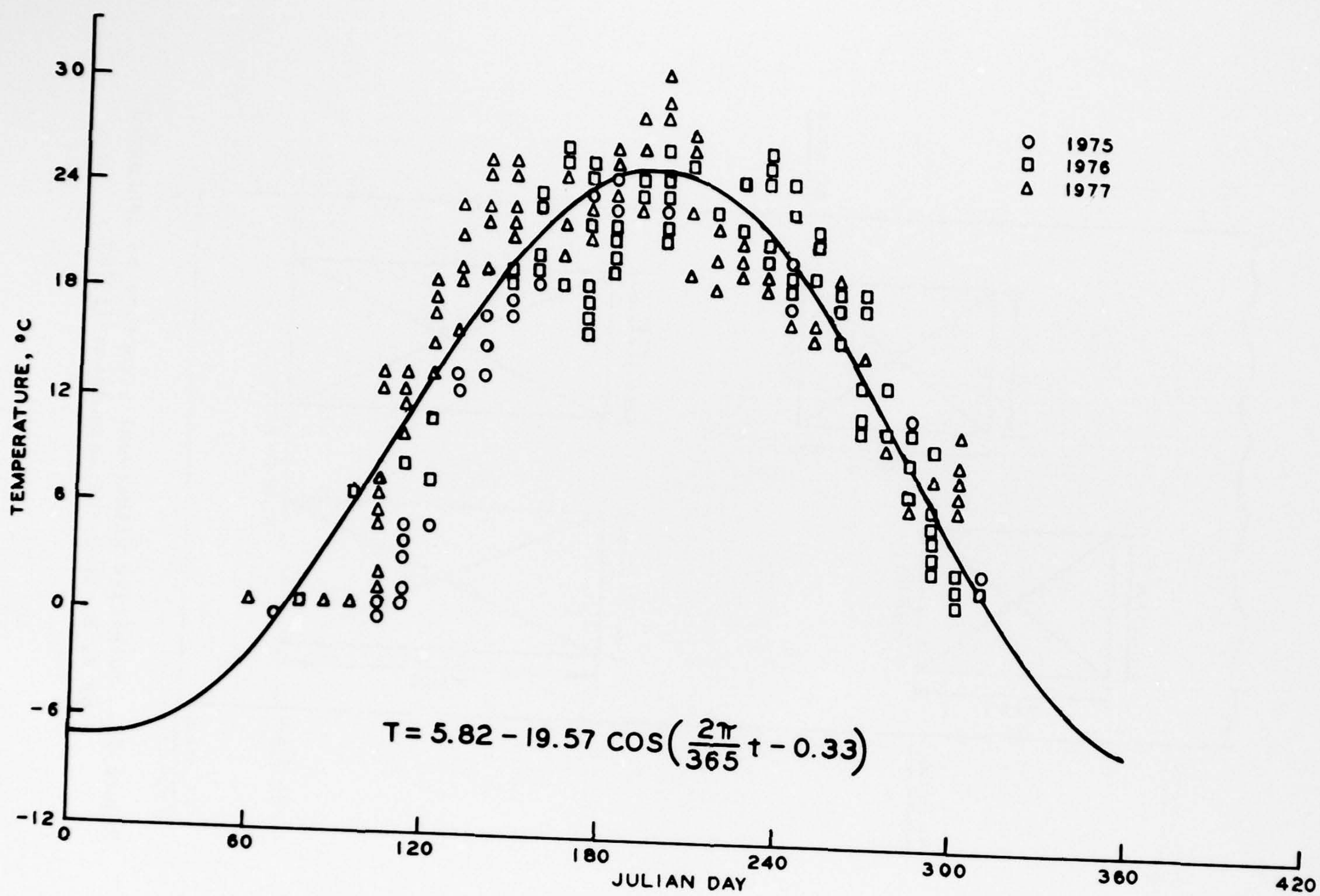


Figure 27. Comparison of downstream temperature objective and temperature data for 1975-1977

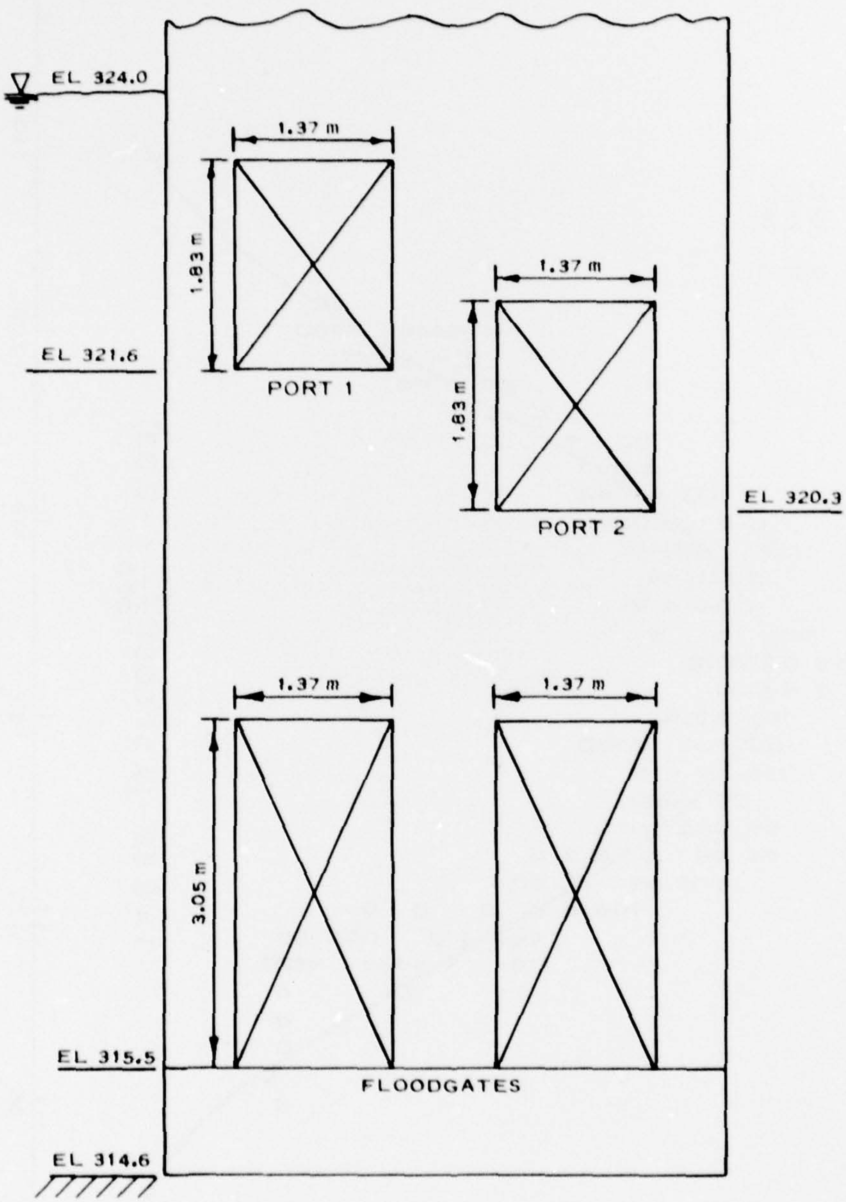


Figure 28. Selective withdrawal structure as conceived by R. W. Beck and Associates (1978)

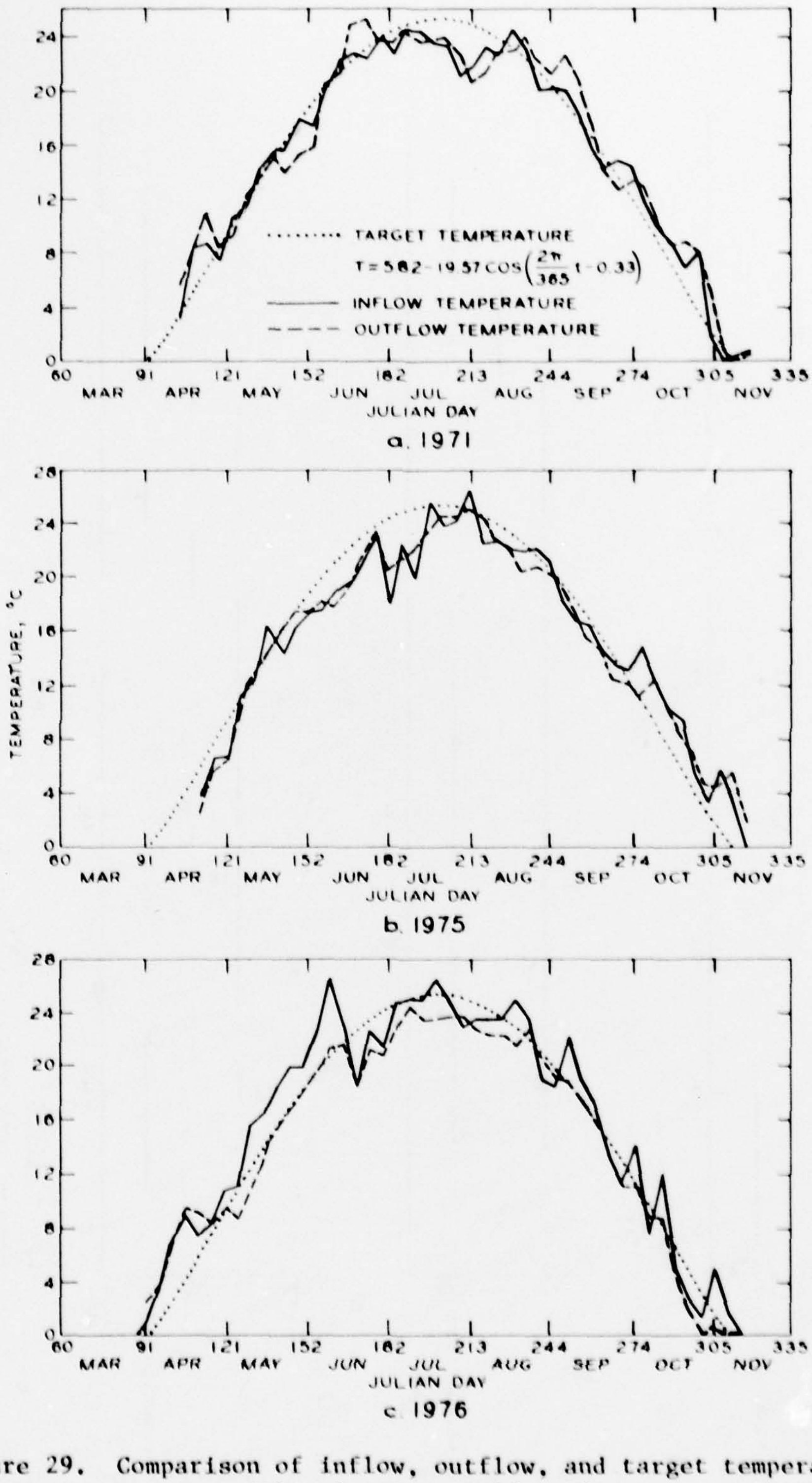


Figure 29. Comparison of inflow, outflow, and target temperatures with selective withdrawal

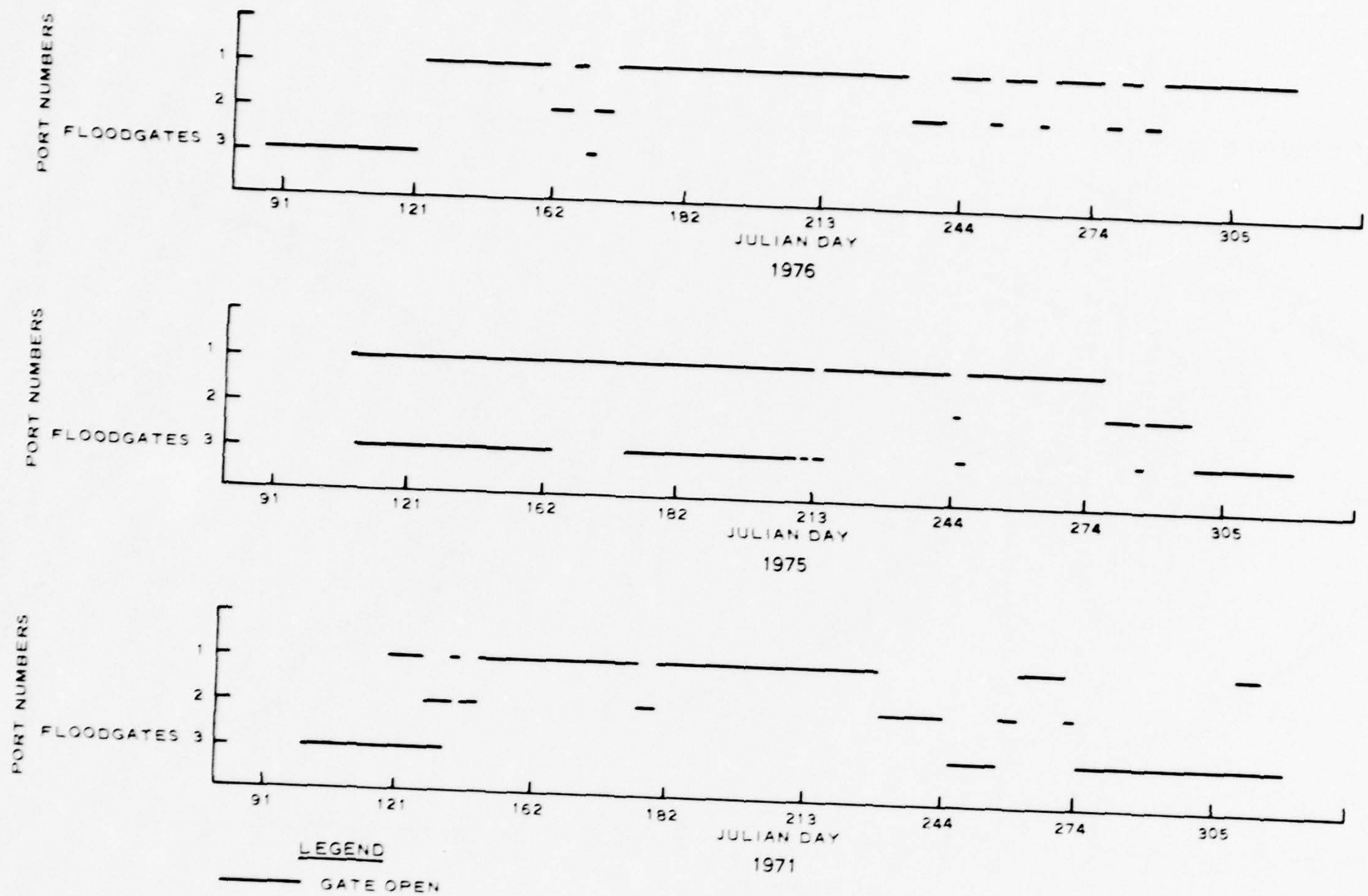


Figure 30. Port operations required to meet natural temperature objective

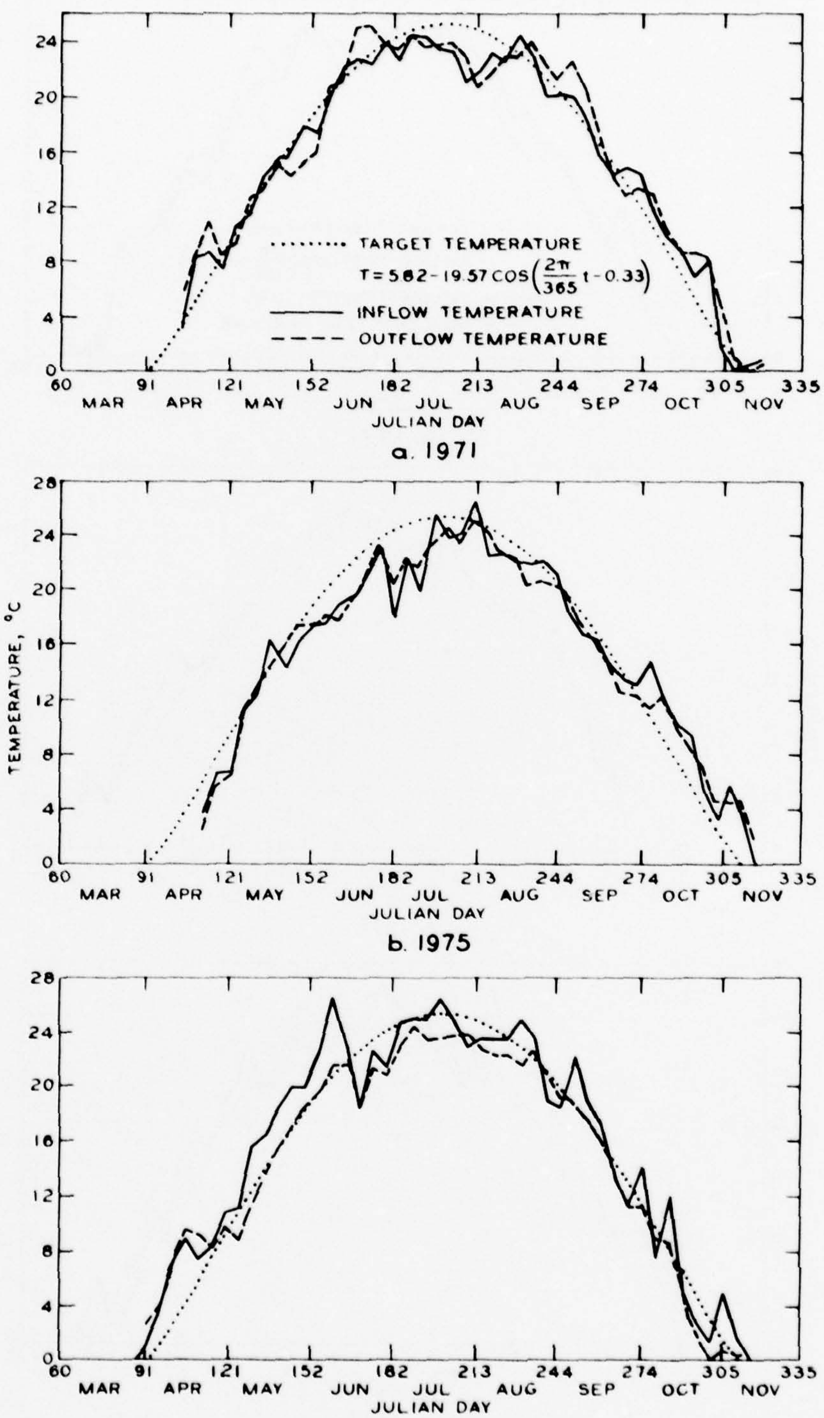


Figure 31. Comparison of inflow, outflow, and target temperatures with surface withdrawal

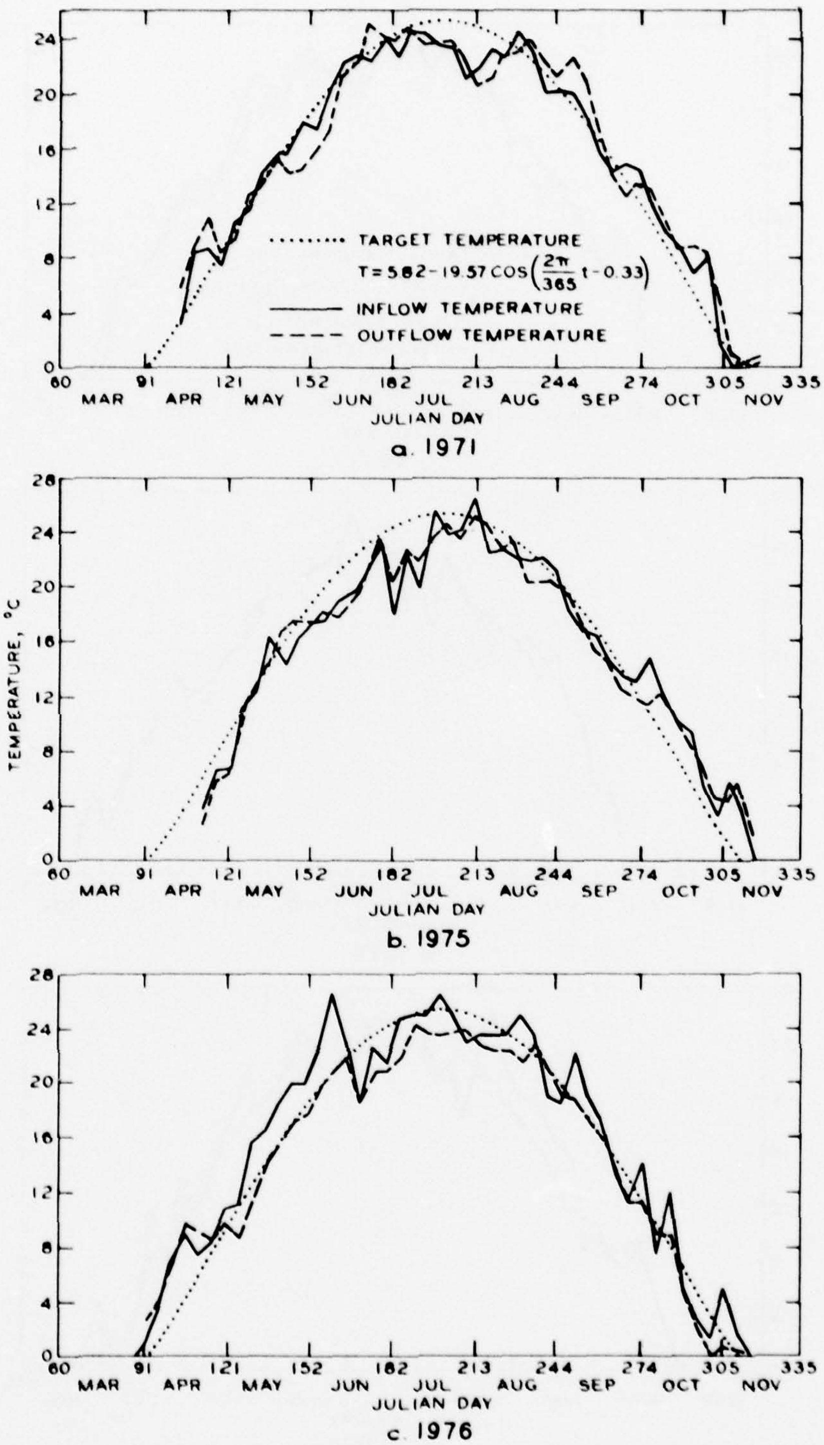


Figure 32. Comparison of inflow, outflow, and target temperatures with bottom withdrawal

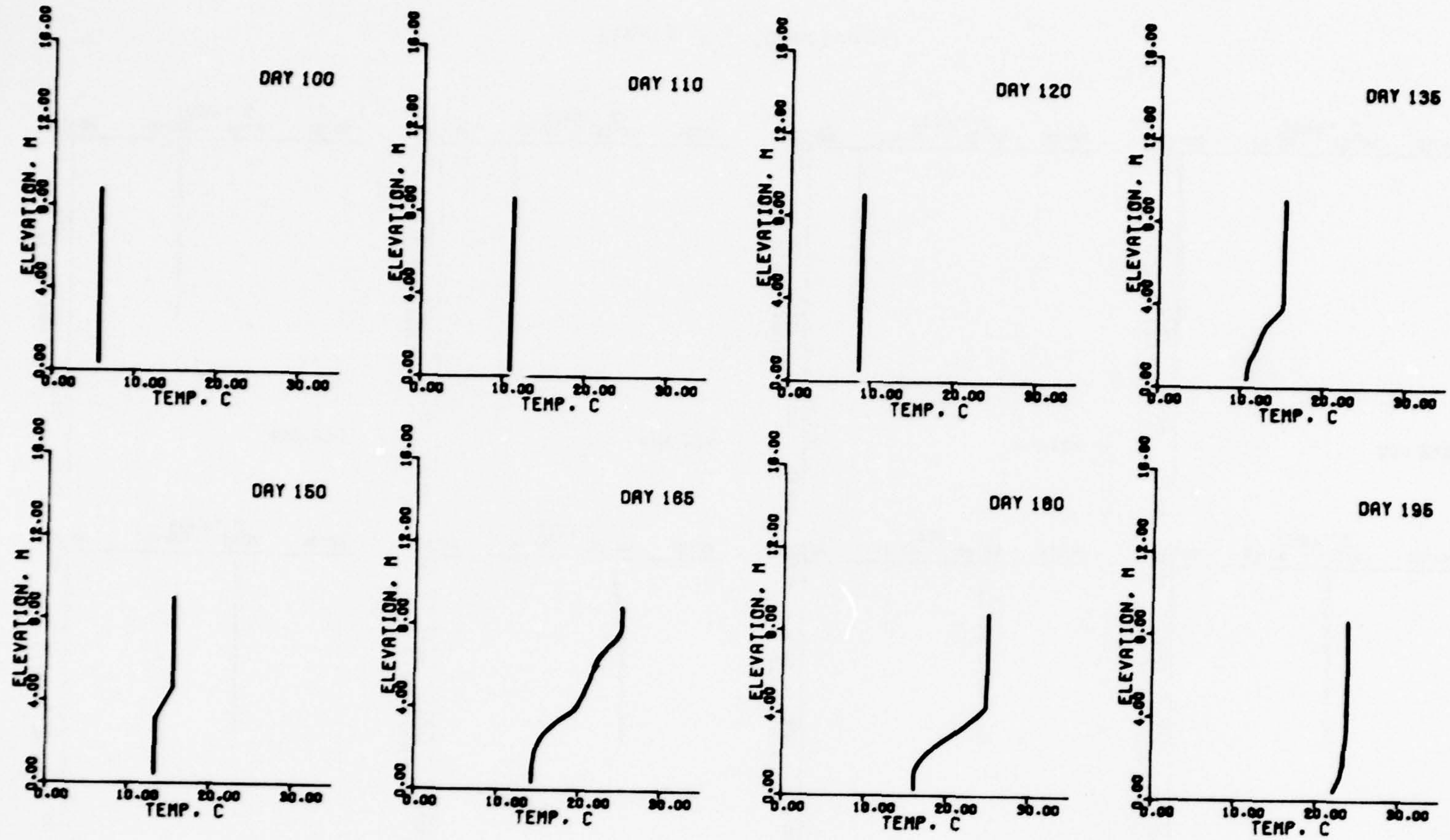


Figure 33. Simulated temperature profiles, Twin Valley Lake, 1971 (Continued)

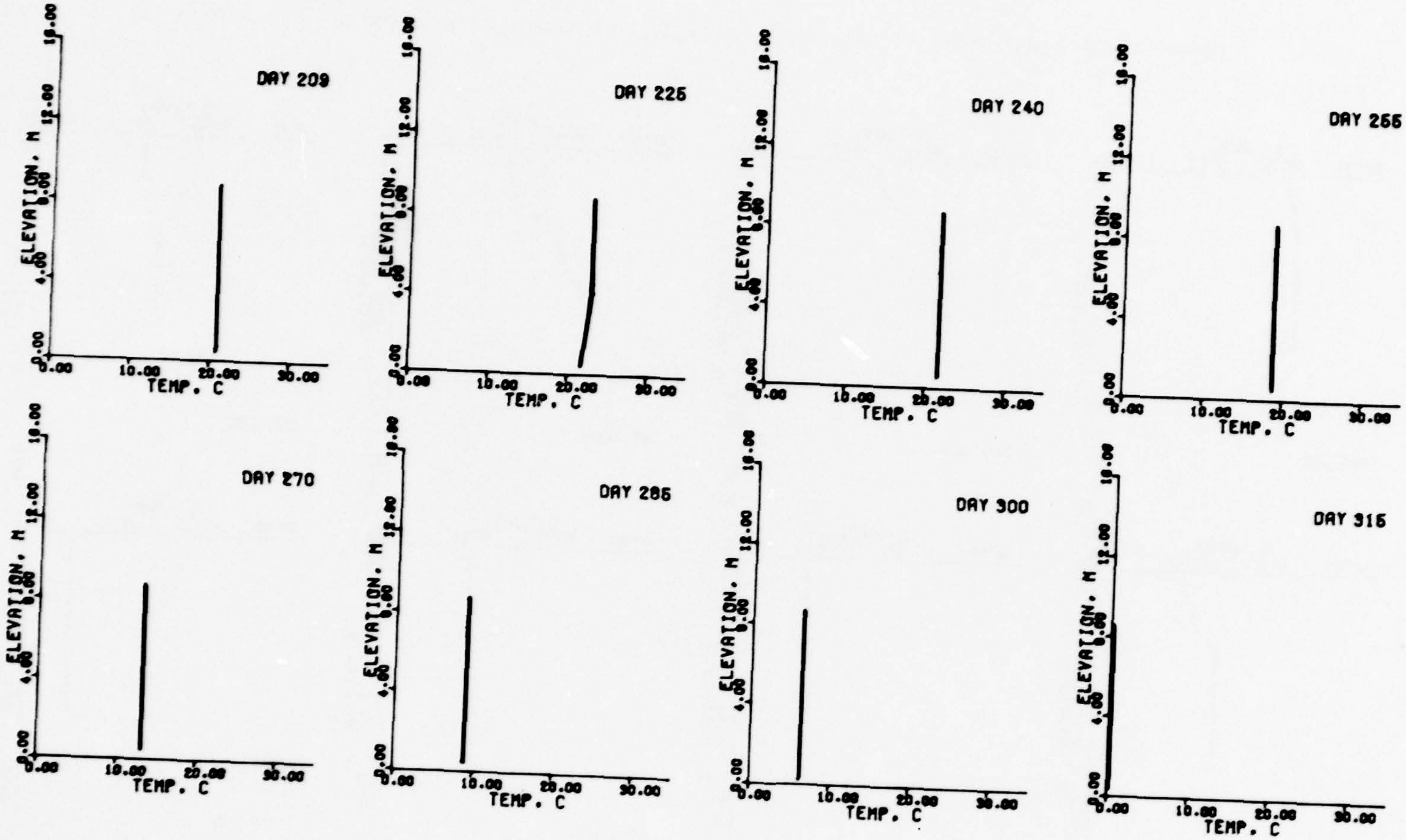


Figure 33. (Concluded)

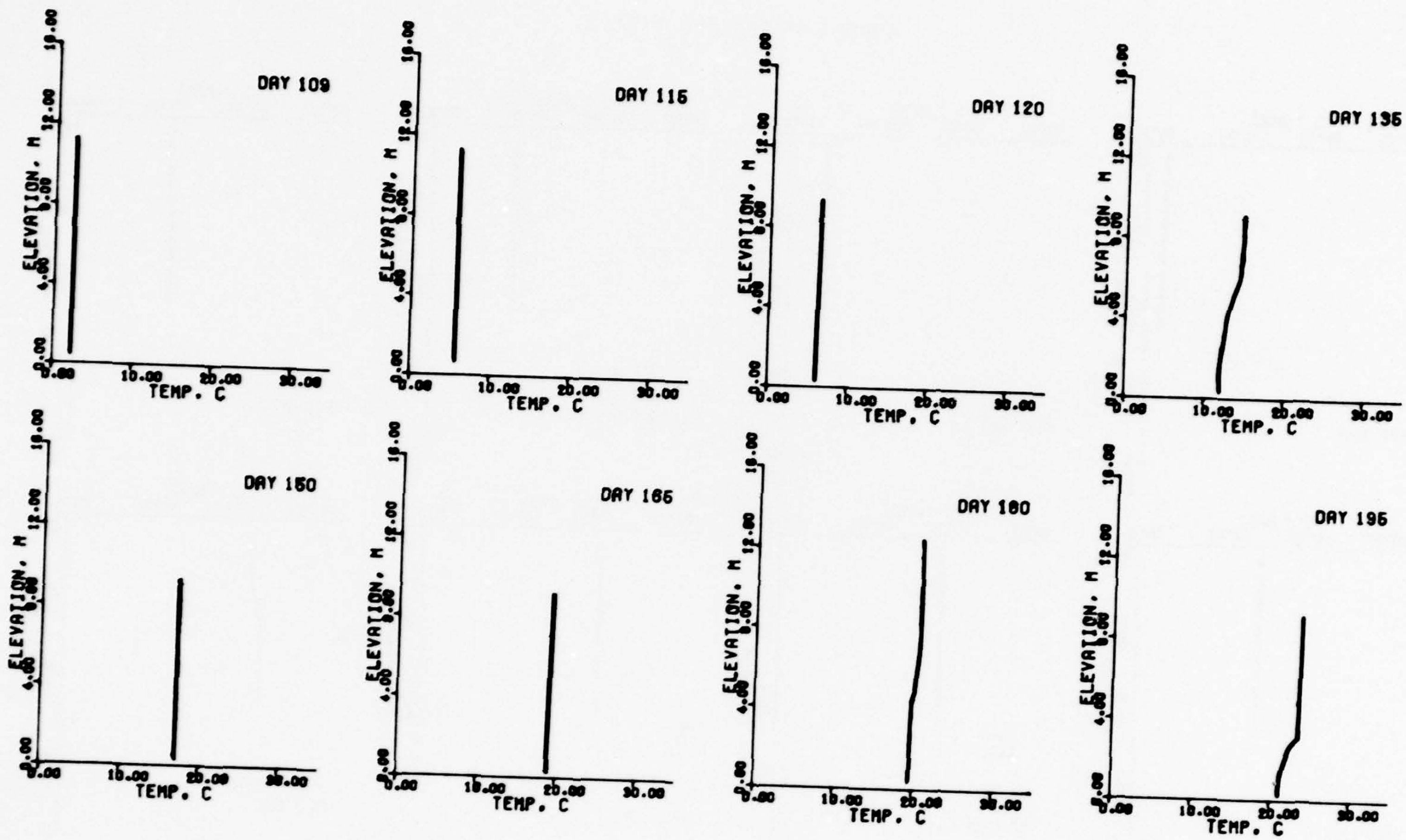


Figure 34. Simulated temperature profiles, Twin Valley Lake, 1975 (Continued)

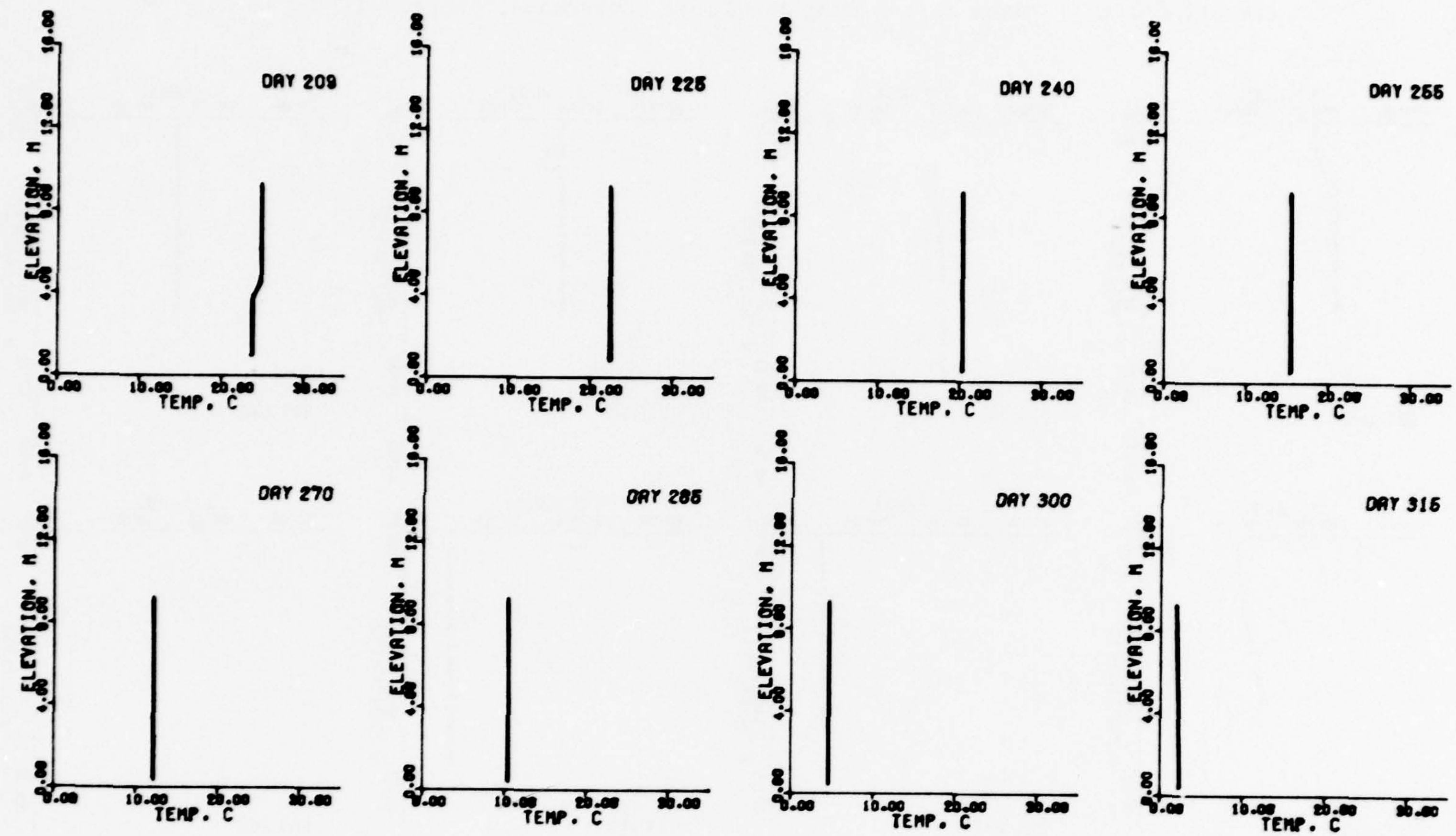


Figure 34. (Concluded)

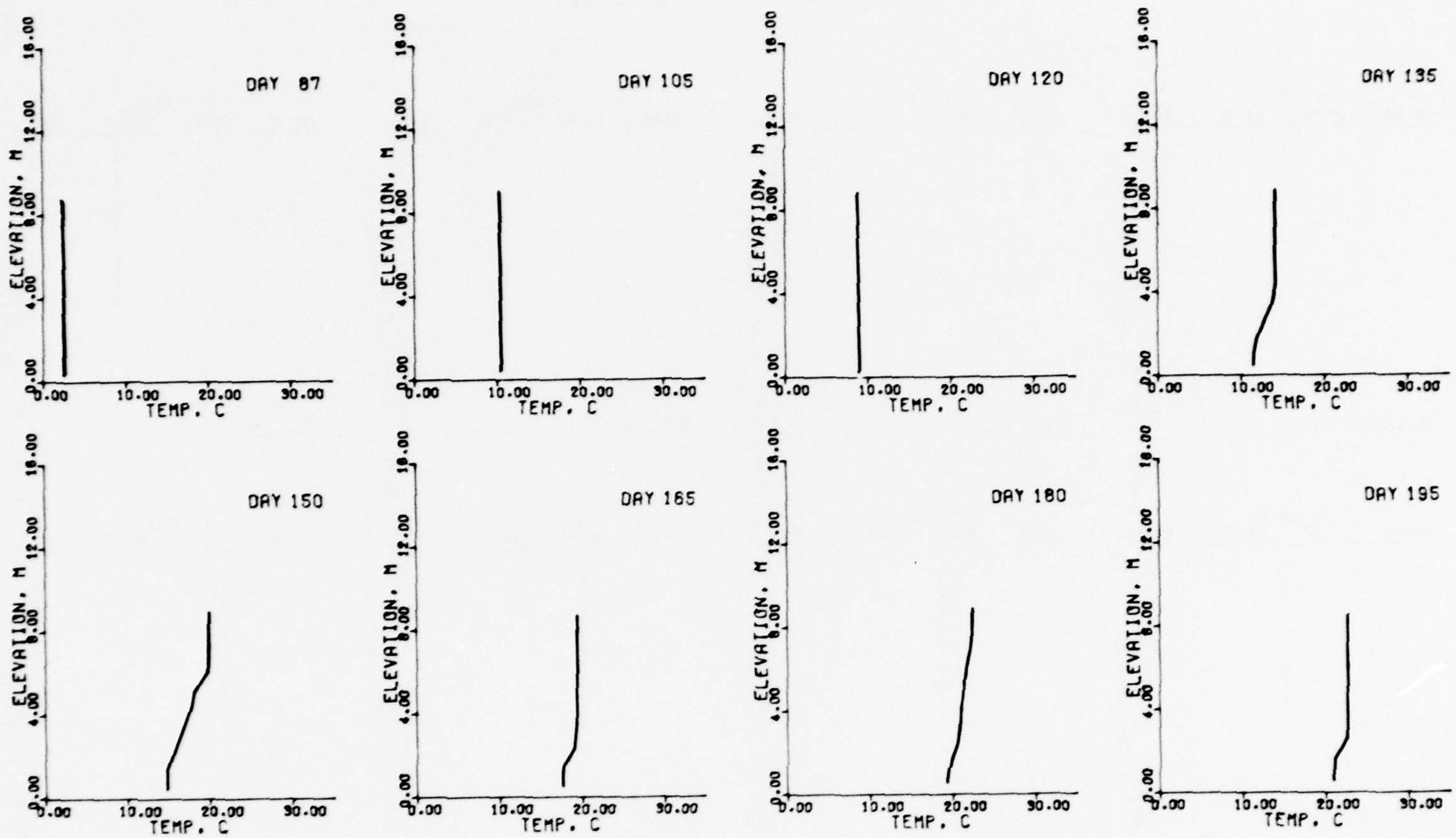


Figure 35. Simulated temperature profiles, Twin Valley Lake, 1976 (Continued)

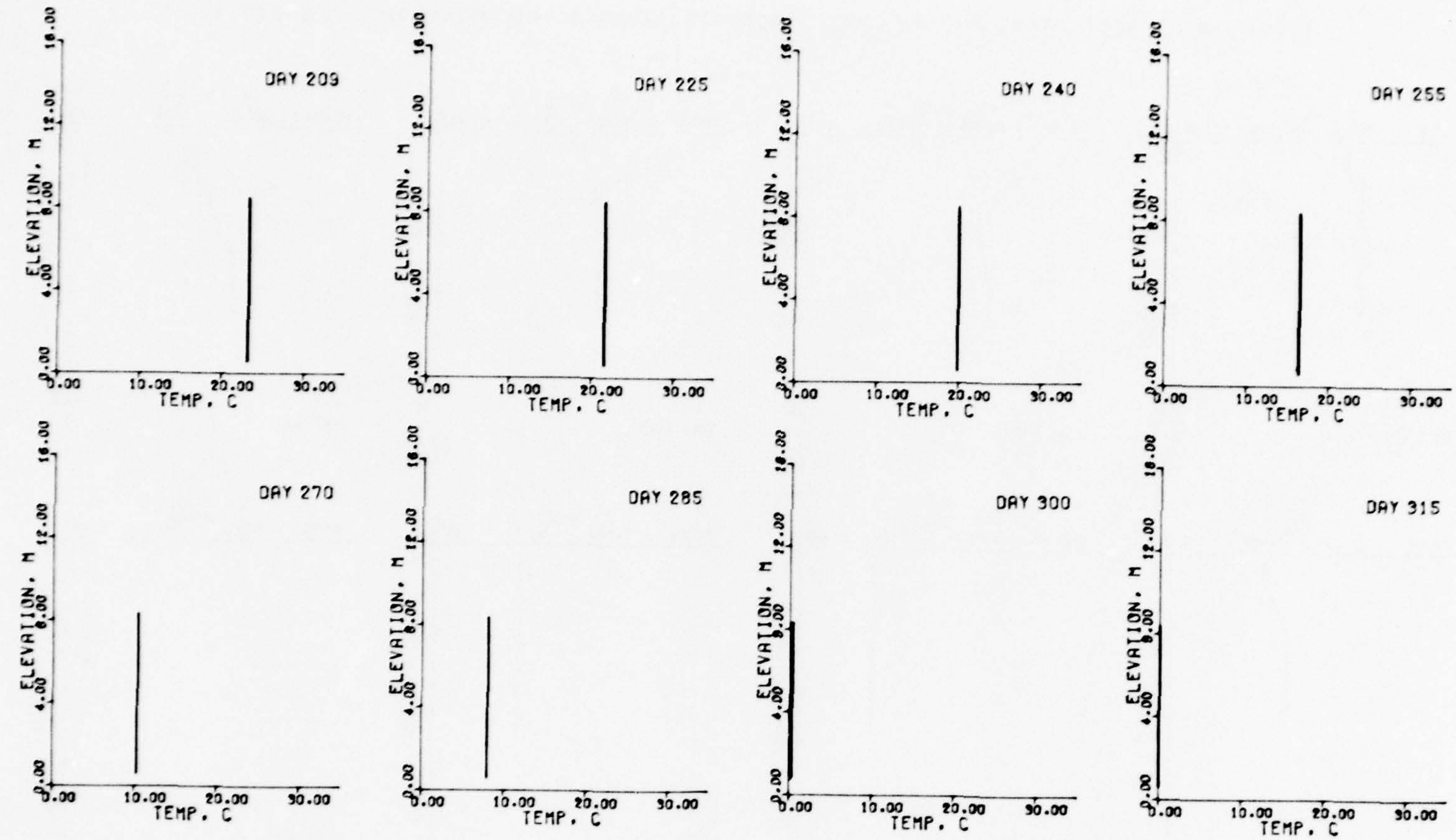
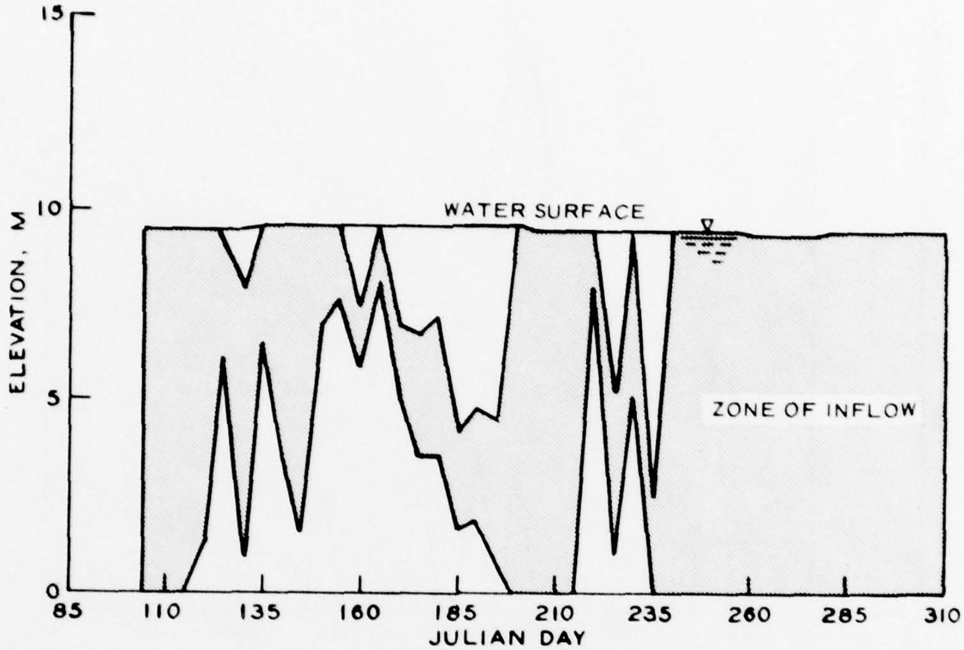
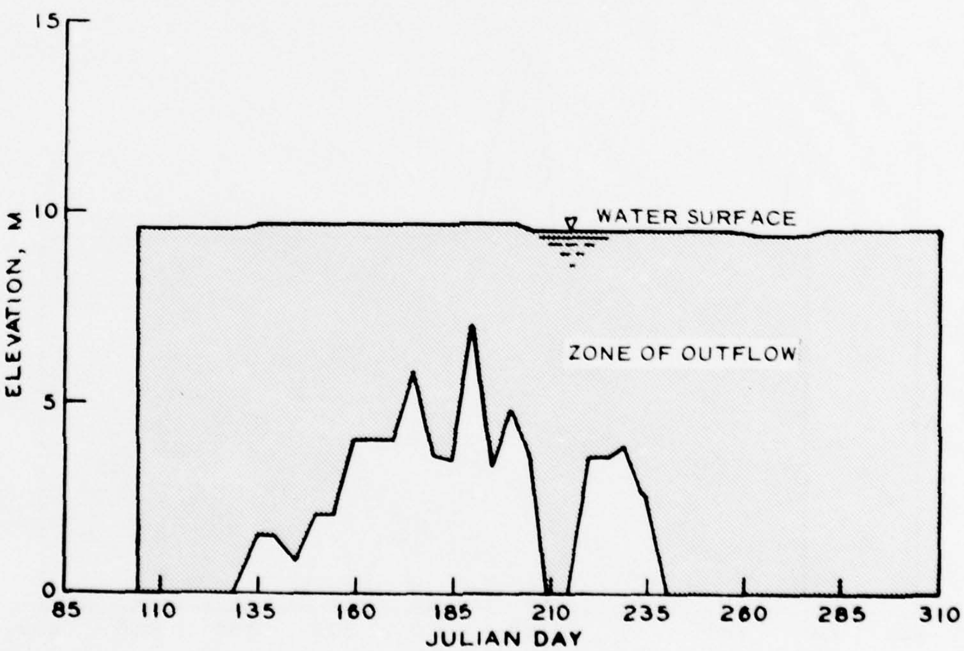


Figure 35. (Concluded)



a. INFLOW



b. OUTFLOW

Figure 36. Zones of inflow and outflow, Twin Valley Lake, 1971

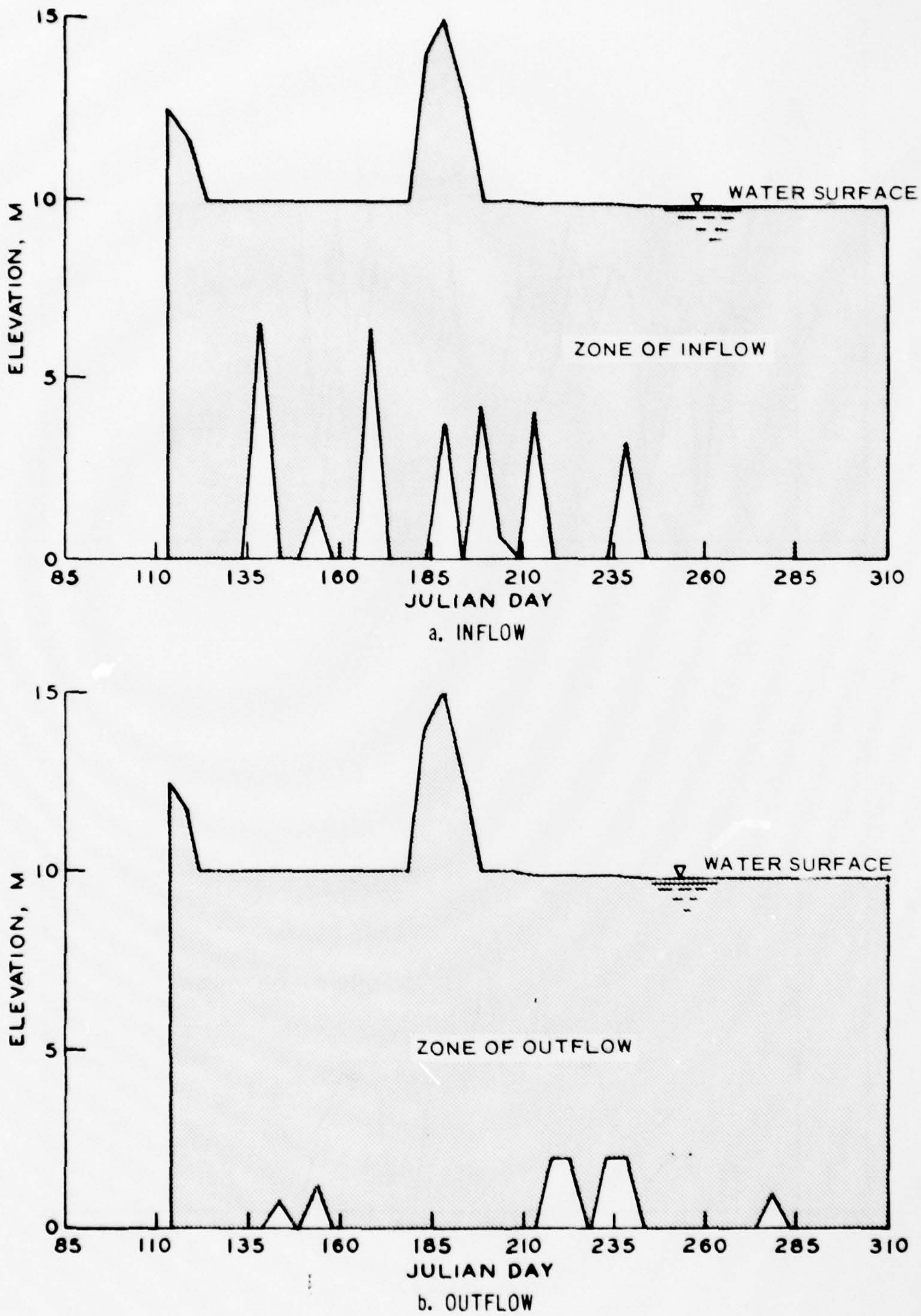


Figure 37. Zones of inflow and outflow, Twin Valley Lake, 1975

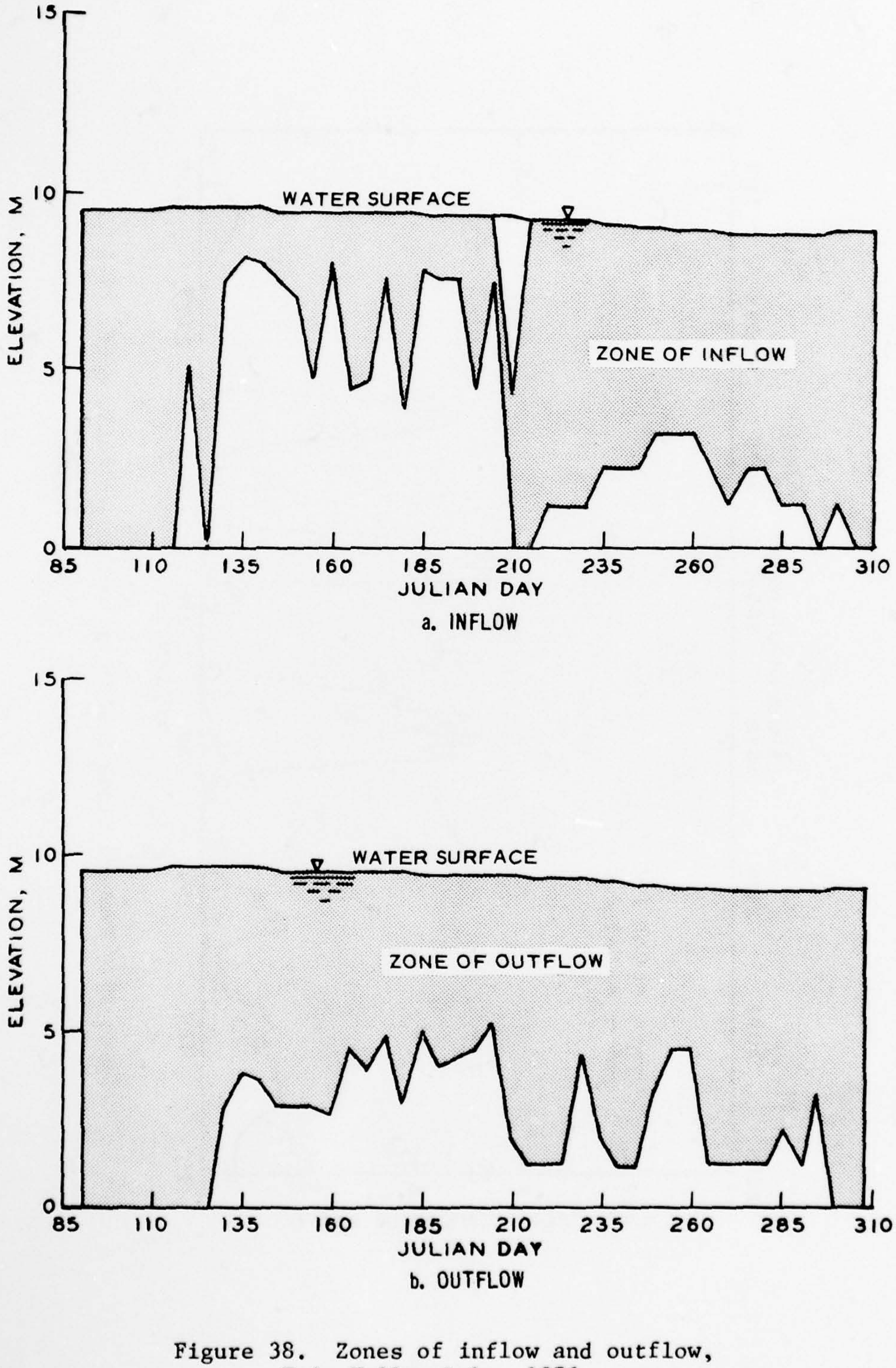


Figure 38. Zones of inflow and outflow,
Twin Valley Lake, 1976

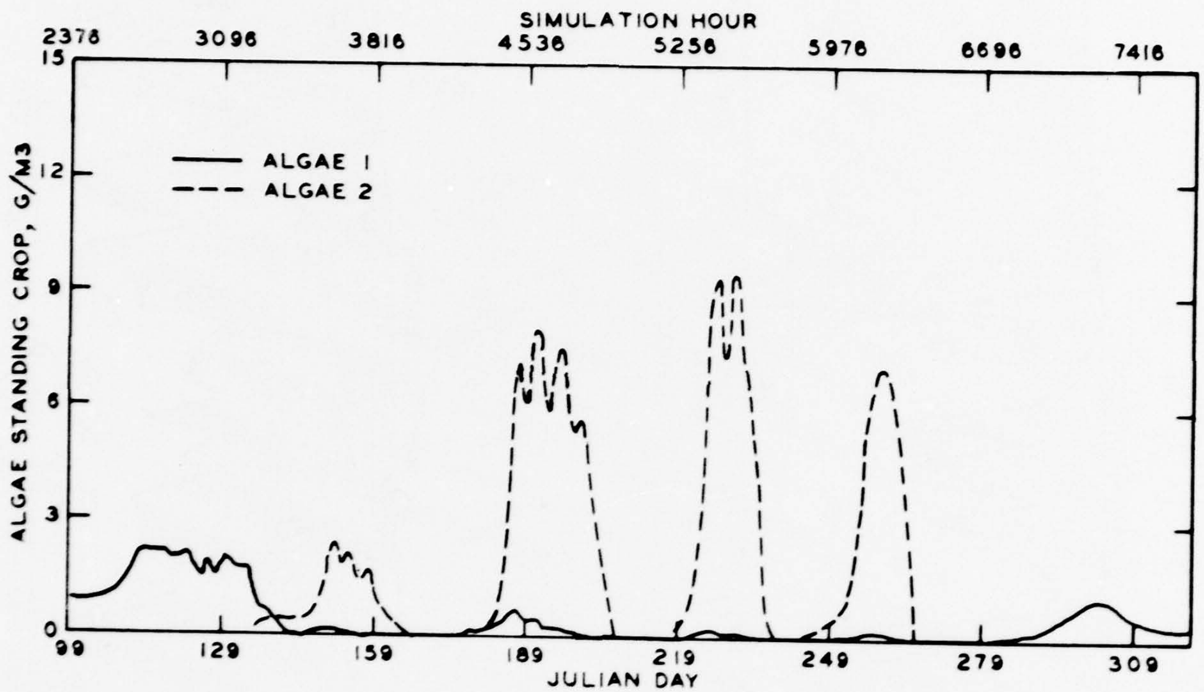


Figure 39. Average algae concentrations in euphotic zone, 1971

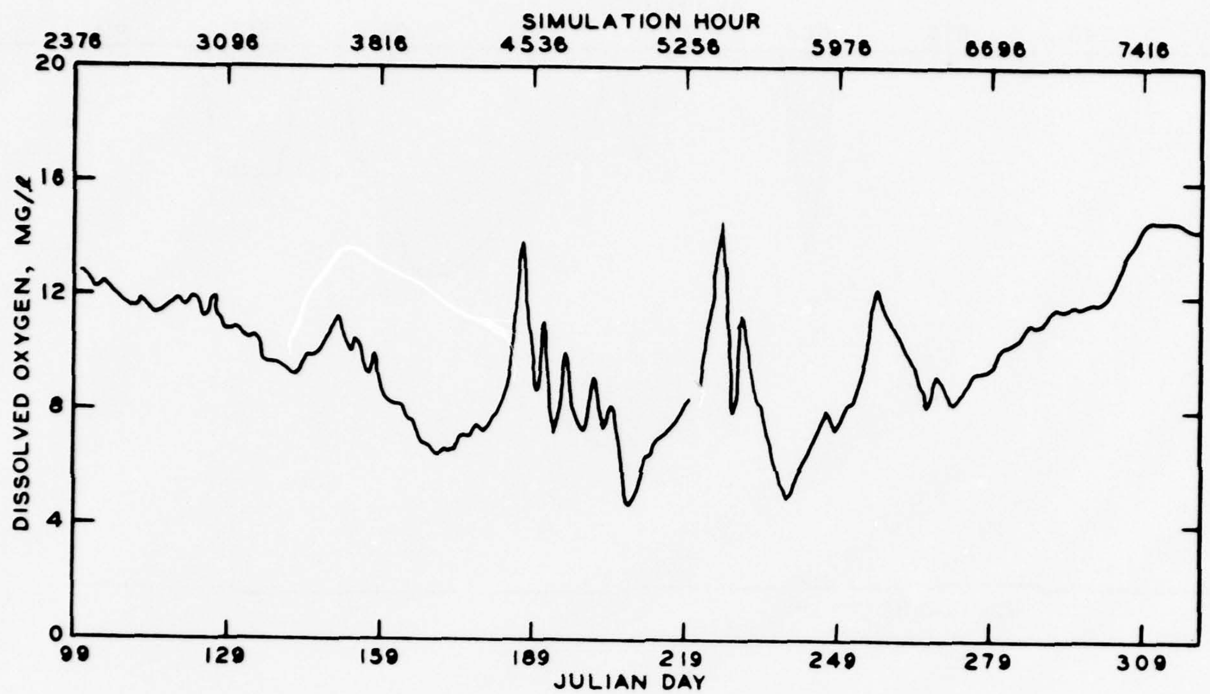


Figure 40. Average DO in surface waters, 1971

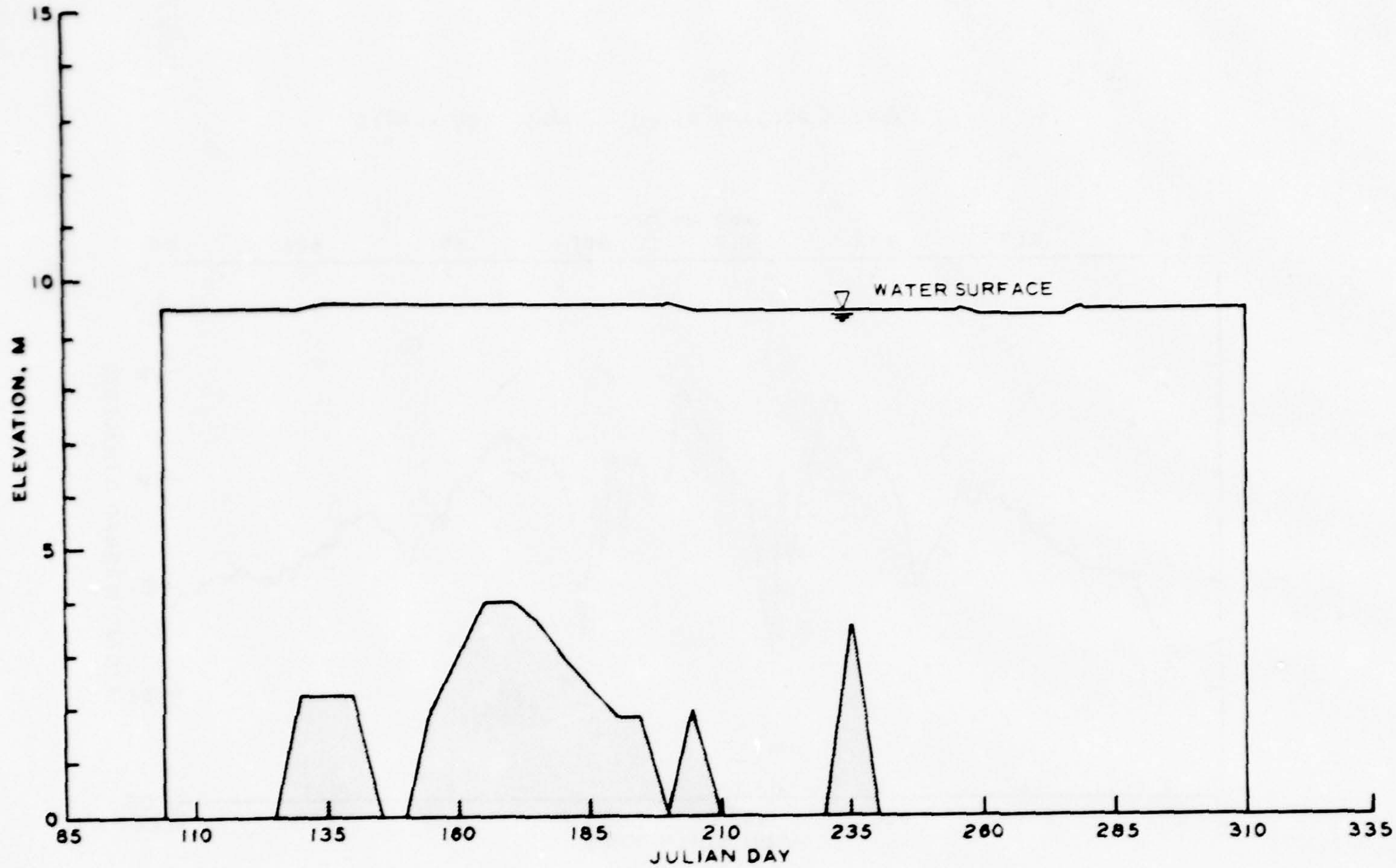


Figure 41. Zone of anoxia (DO < 2 mg/ℓ), 1971

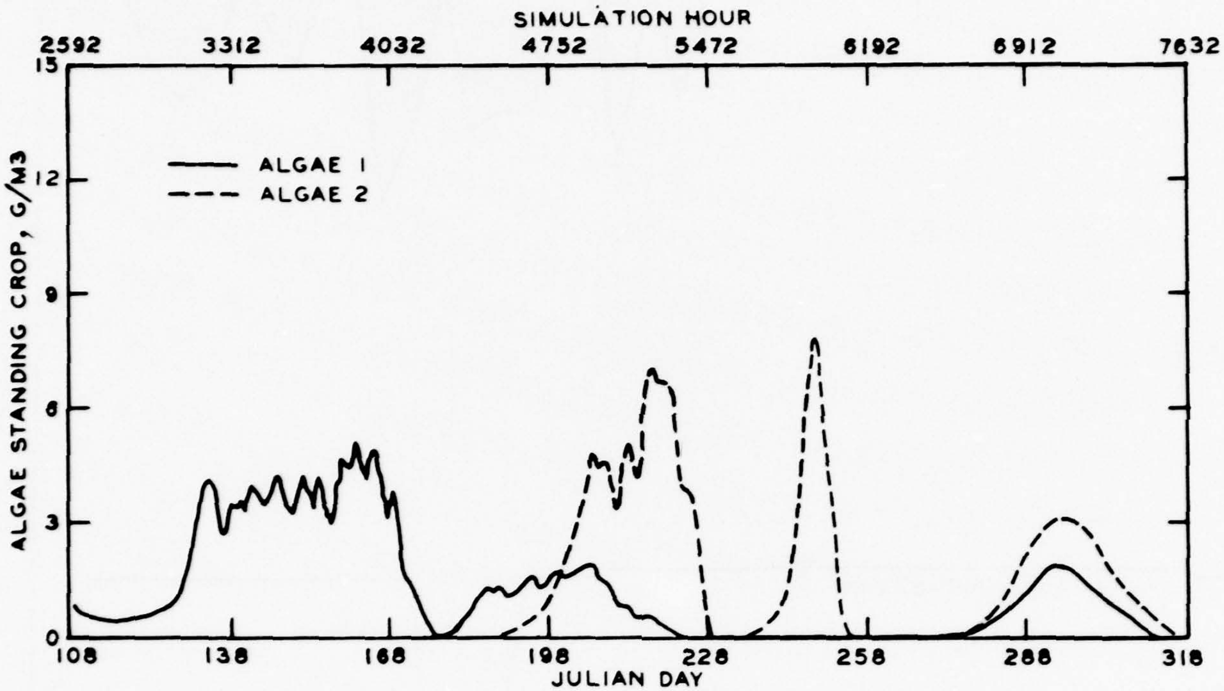


Figure 42. Average algae concentrations in euphotic zone, 1975

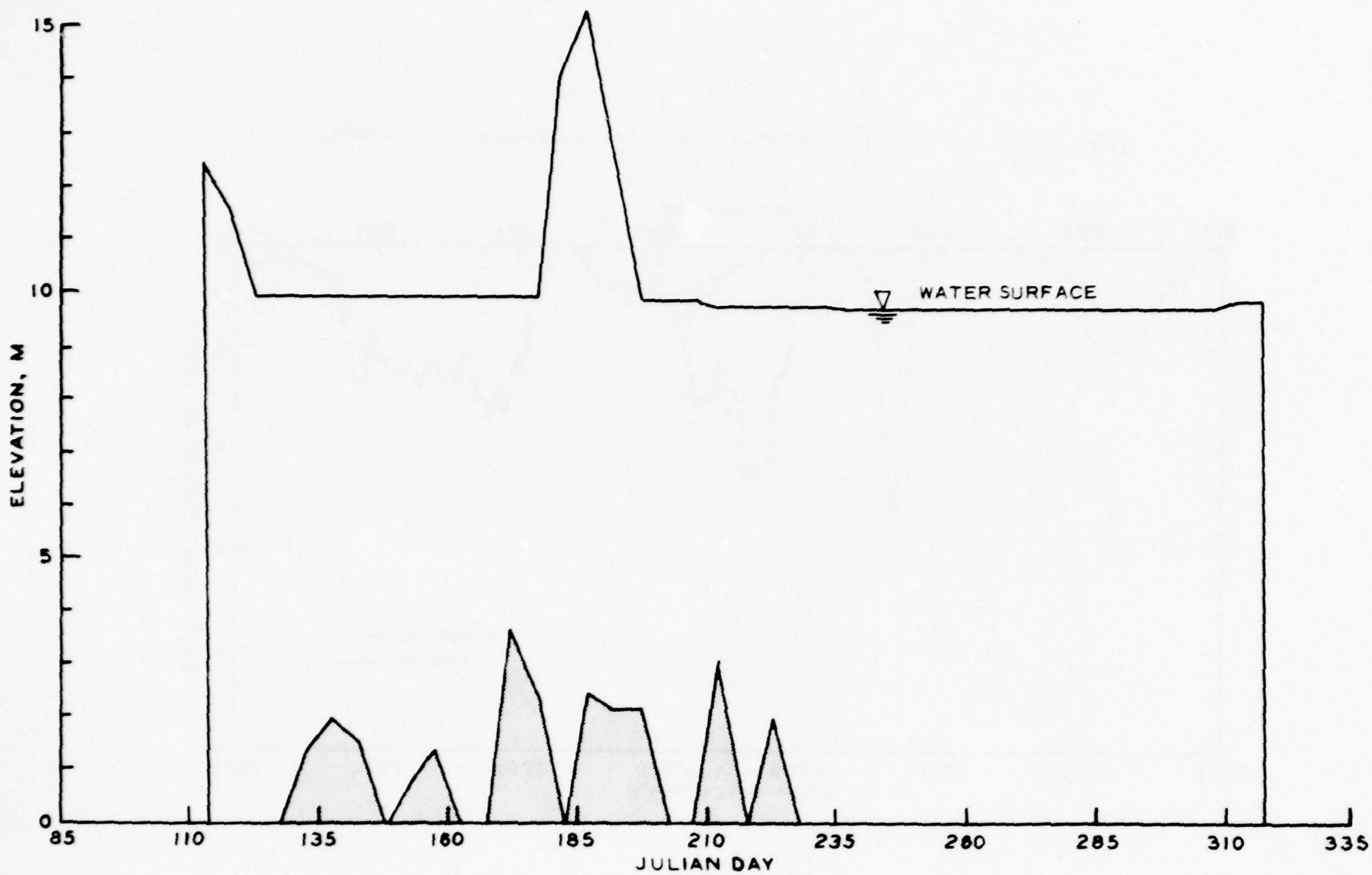


Figure 43. Zone of anoxia (DO < 2 mg/ℓ), 1975

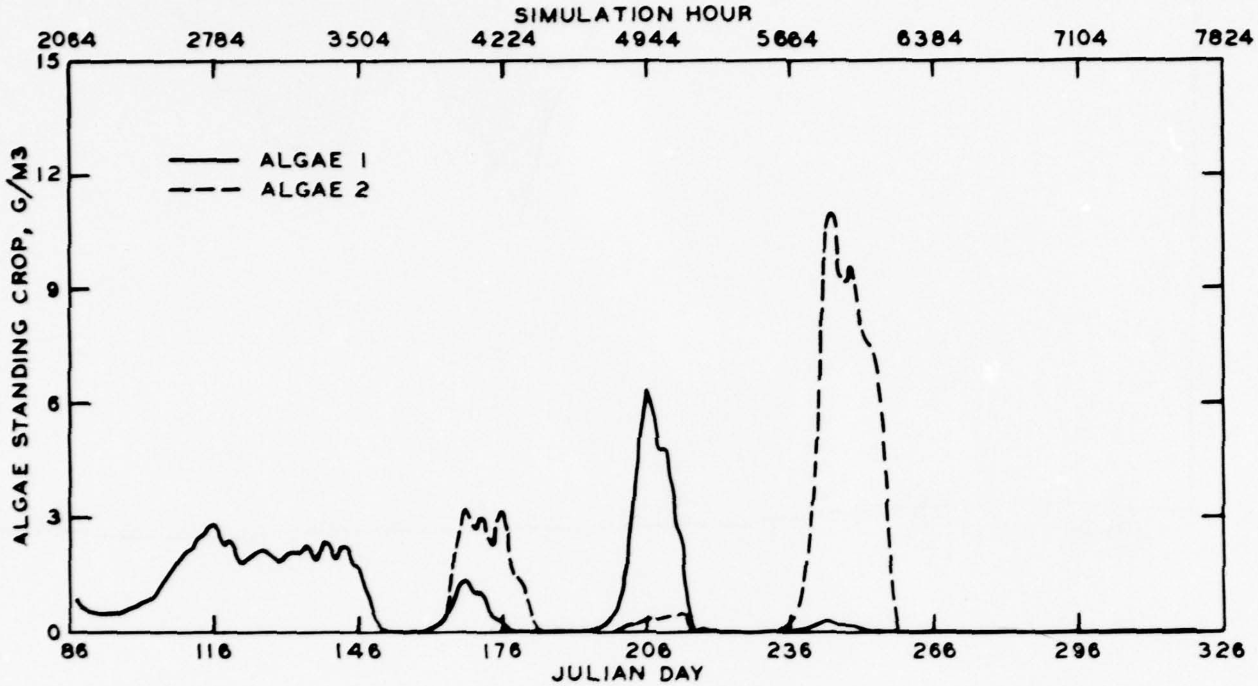


Figure 44. Average algae concentration in euphotic zone, 1976

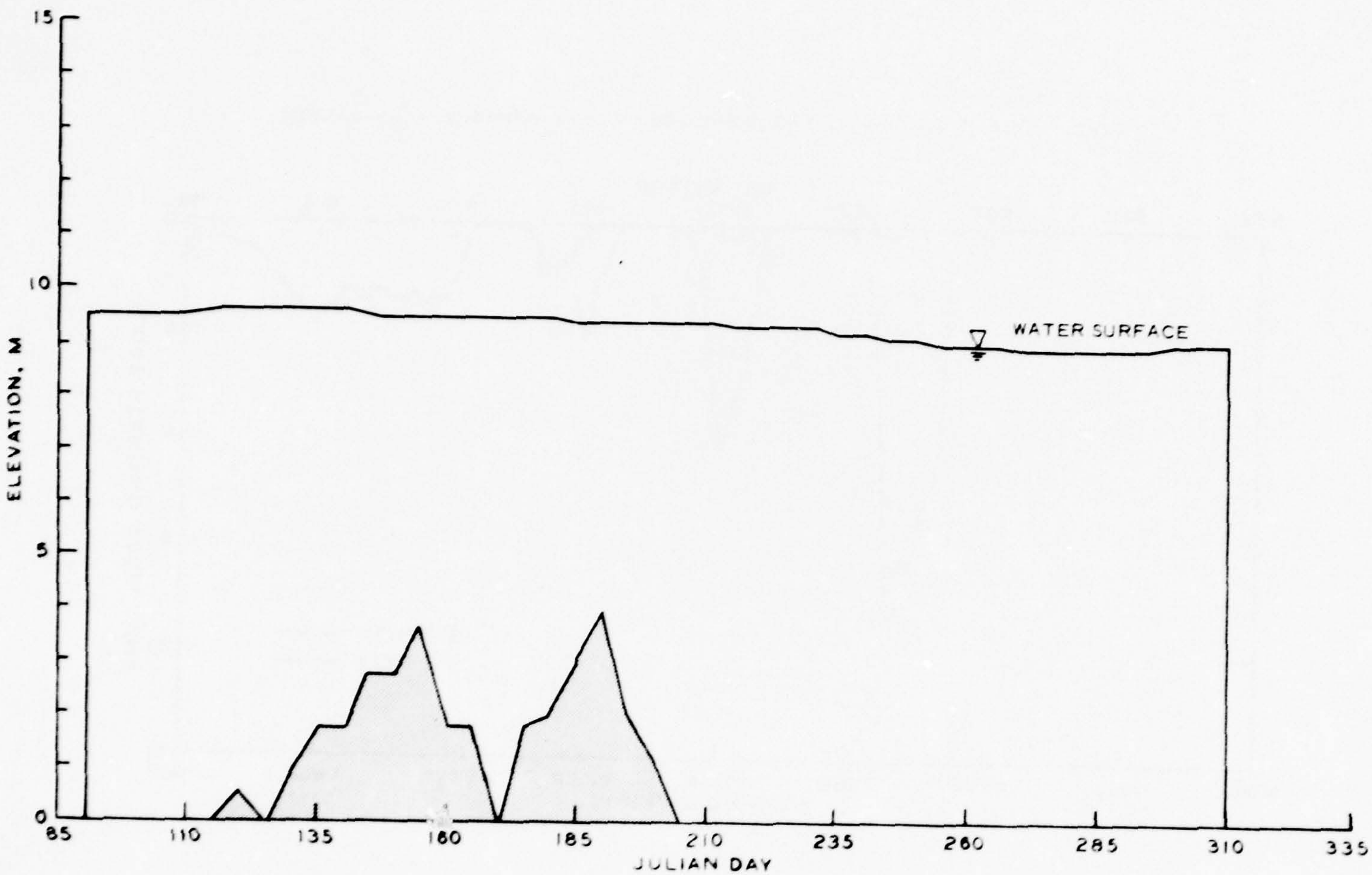


Figure 45. Zone of anoxia (DO < 2 mg/ℓ), 1976

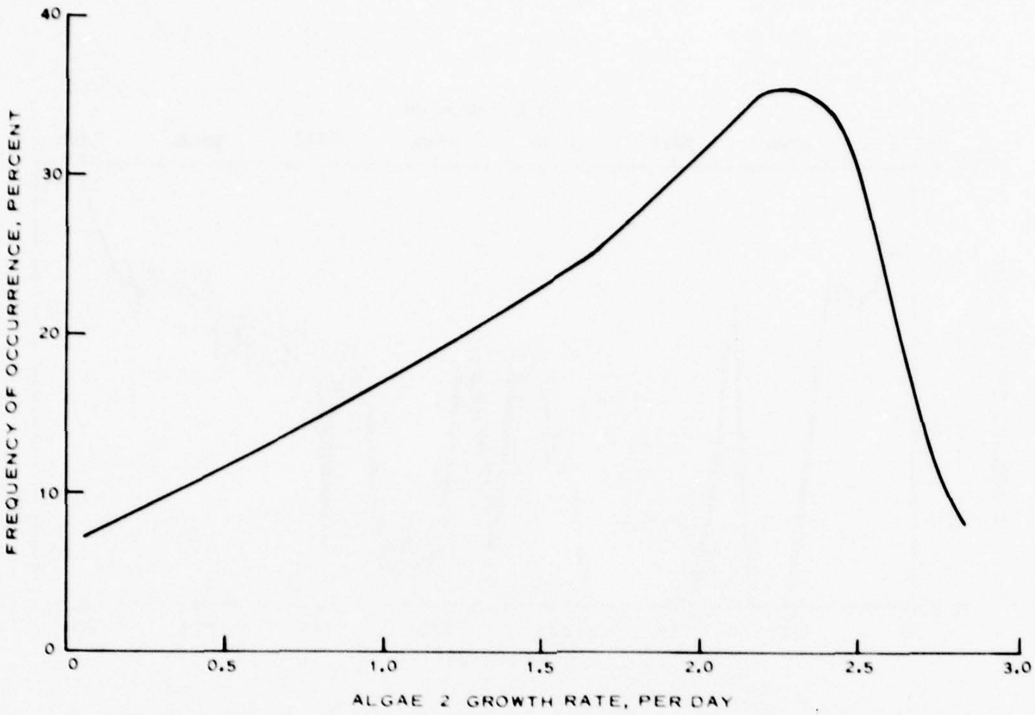


Figure 46. Distribution of ALGAE 2 growth rates

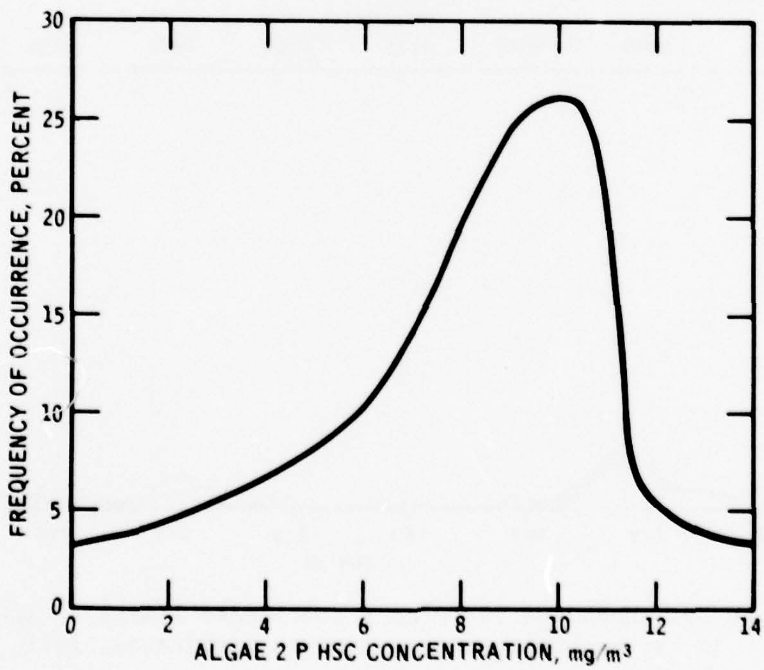


Figure 47. Distribution of ALGAE 2 phosphorus half-saturation coefficients (HSC)

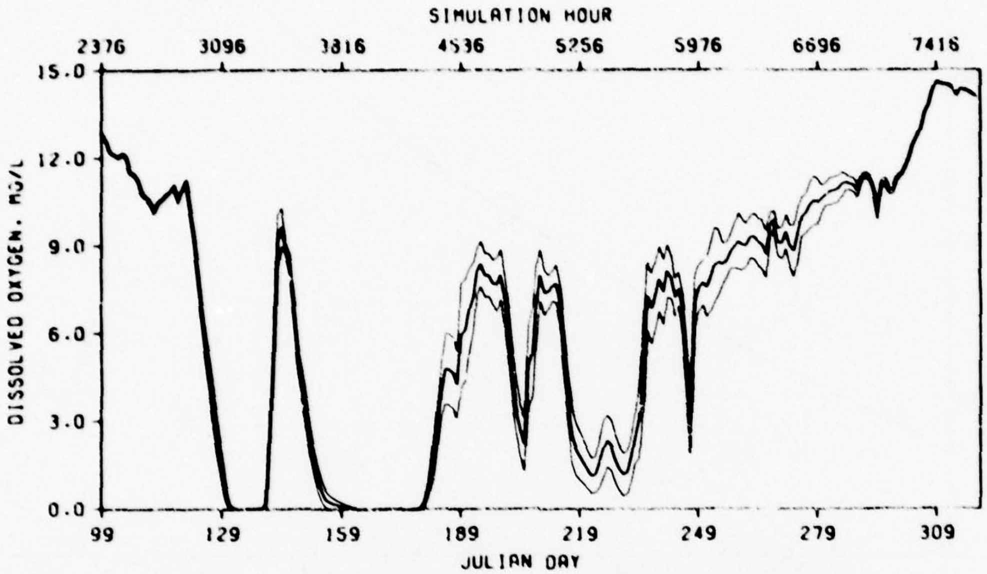


Figure 48. Mean and 95 percent confidence interval of DO in the bottom metre, selective withdrawal, 1971

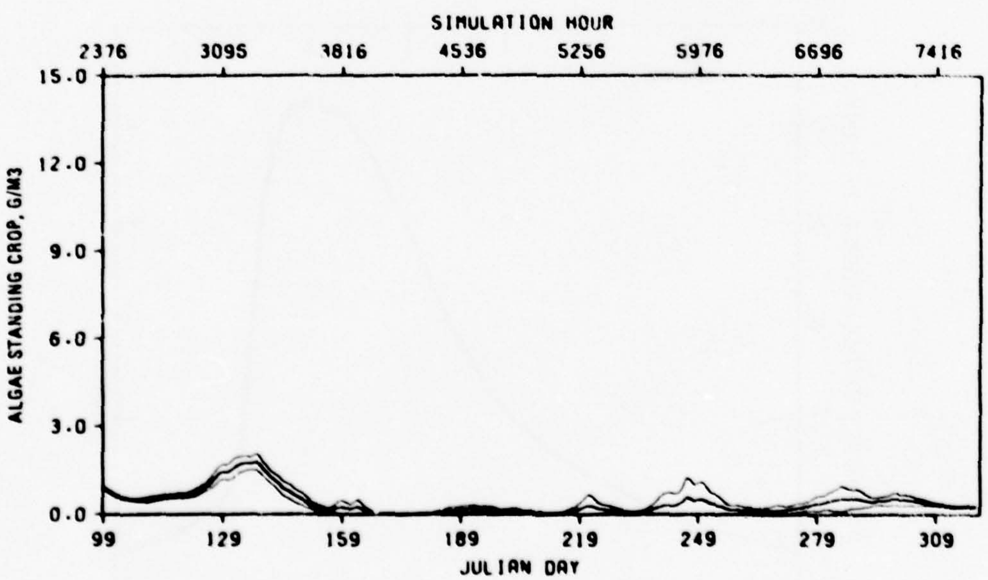


Figure 49. Mean and 95 percent confidence limits of ALGAE 1 in the euphotic zone, selective withdrawal, 1971

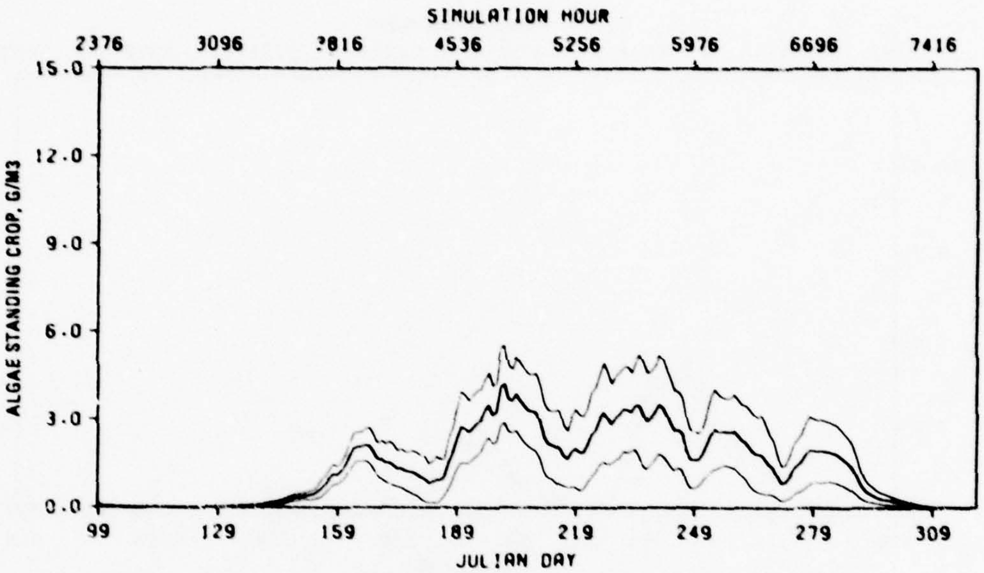


Figure 50. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, selective withdrawal, 1971

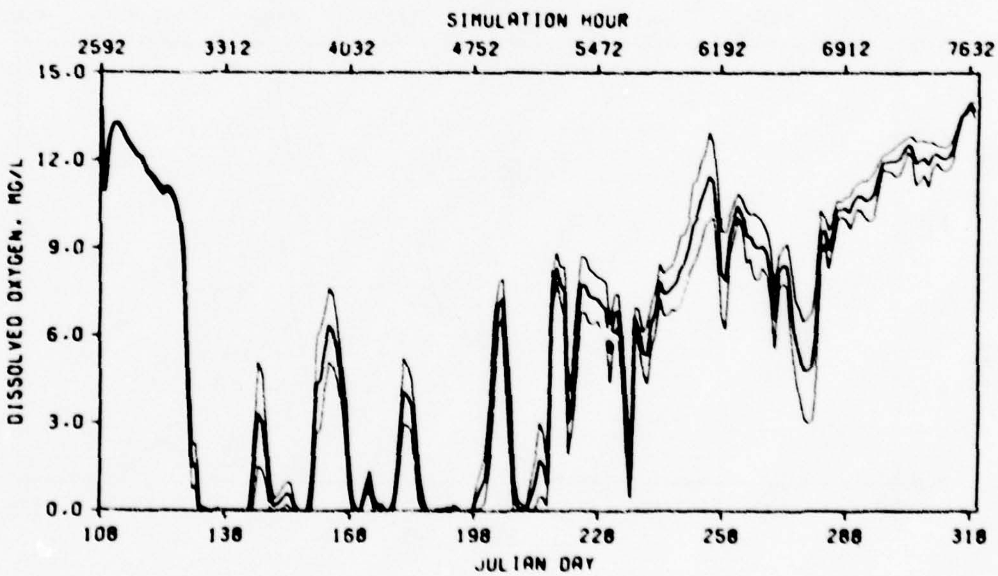


Figure 51. Mean and 95 percent confidence interval of DO in the bottom metre, selective withdrawal, 1975

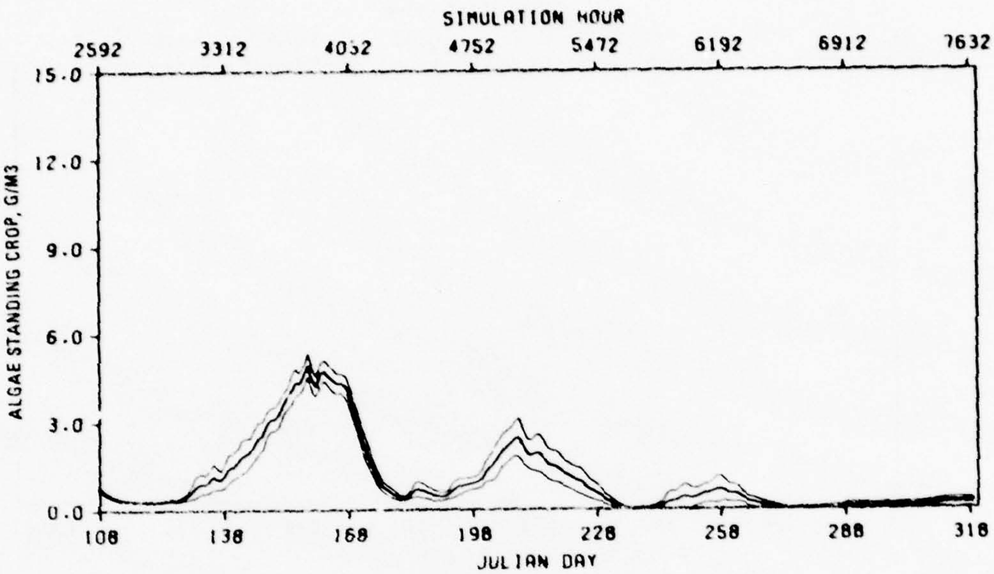


Figure 52. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, selective withdrawal, 1975

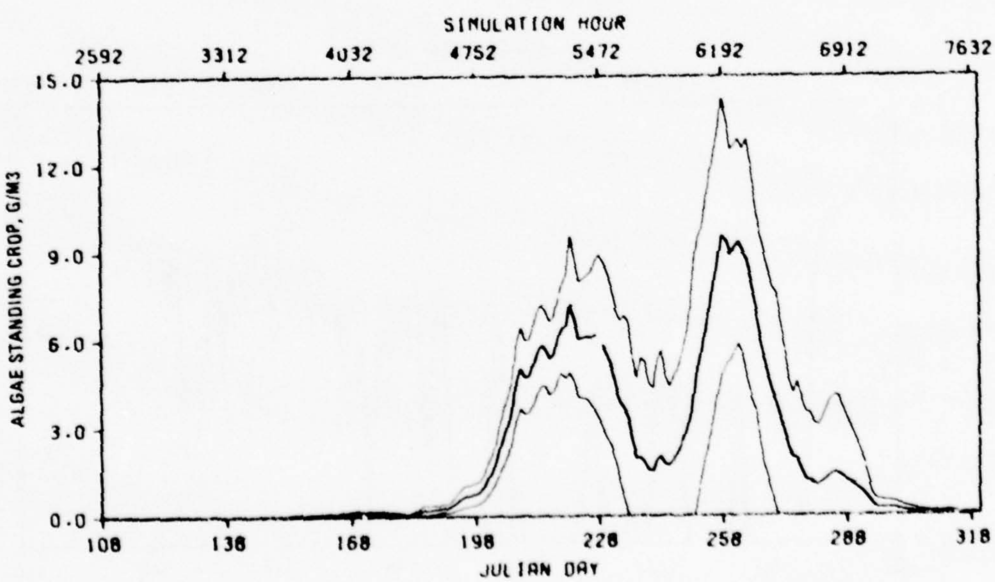


Figure 53. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, selective withdrawal, 1975

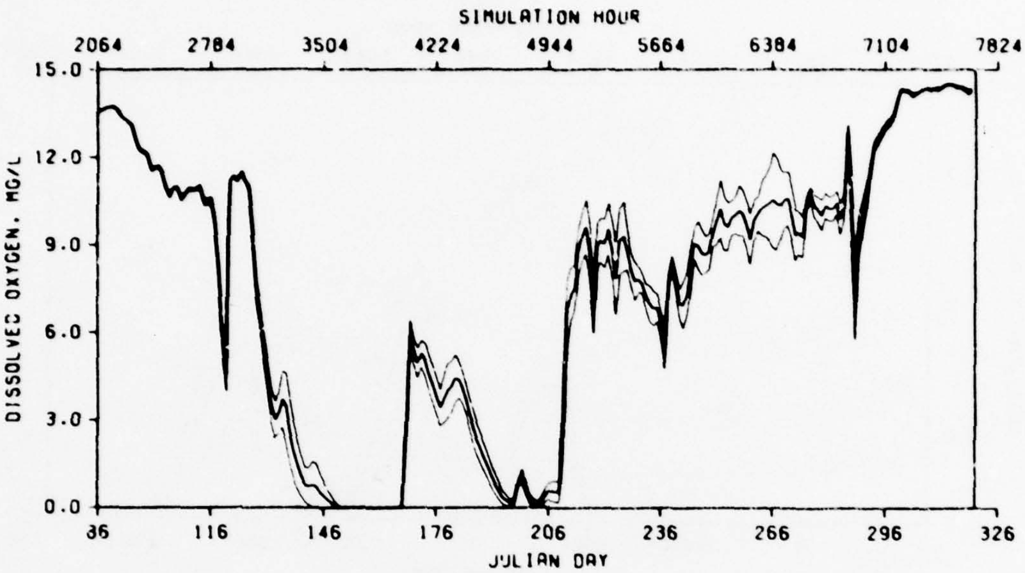


Figure 54. Mean and 95 percent confidence interval of DO in the bottom metre, selective withdrawal, 1976

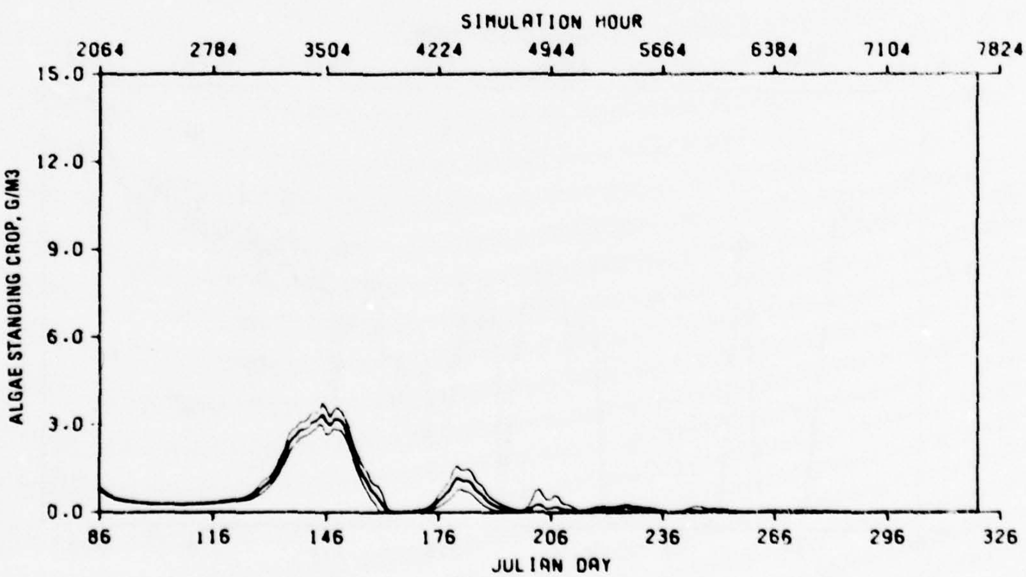


Figure 55. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, selective withdrawal, 1976

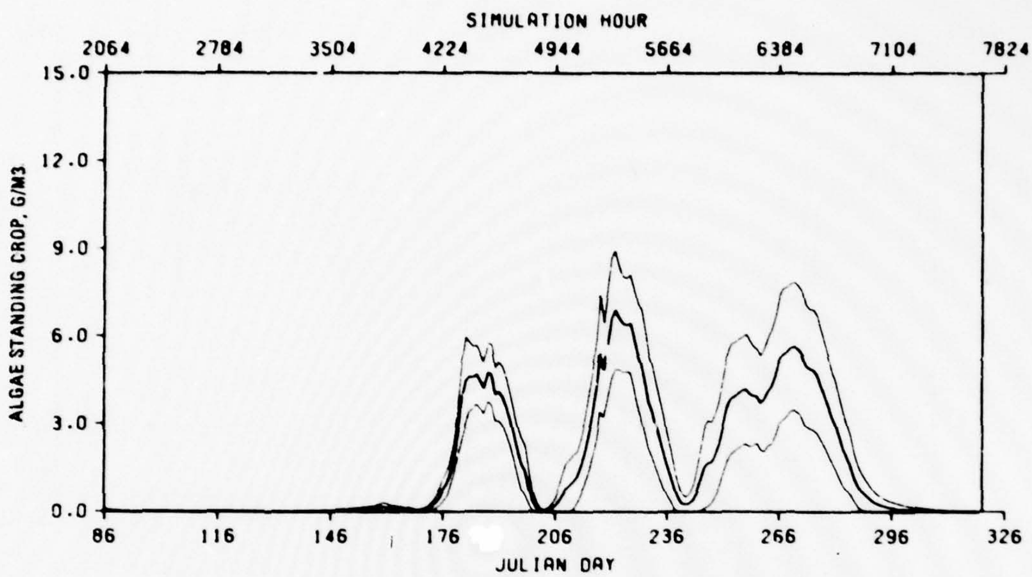


Figure 56. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, selective withdrawal, 1976

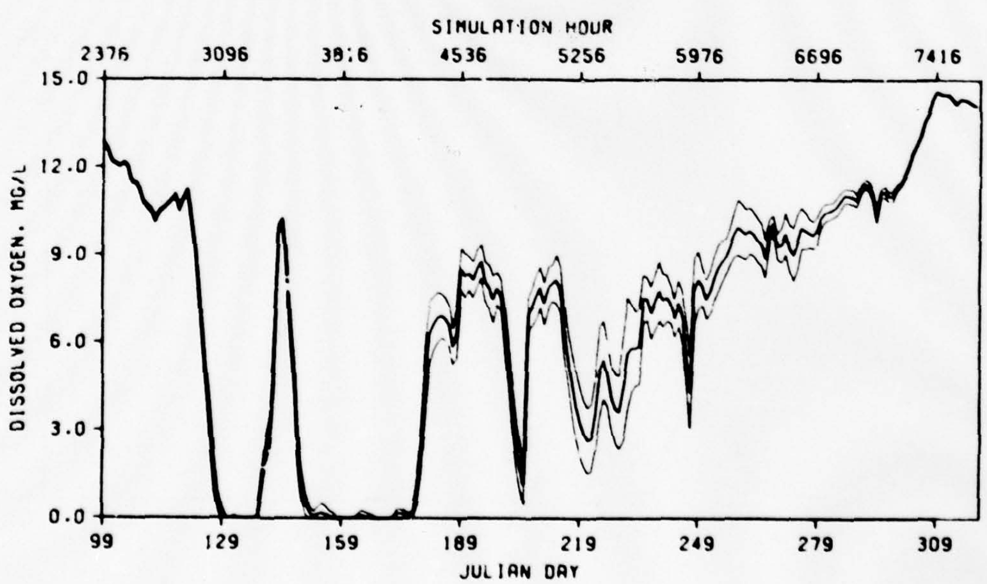


Figure 57. Mean and 95 percent confidence interval of DO in the bottom metre, bottom withdrawal, 1971

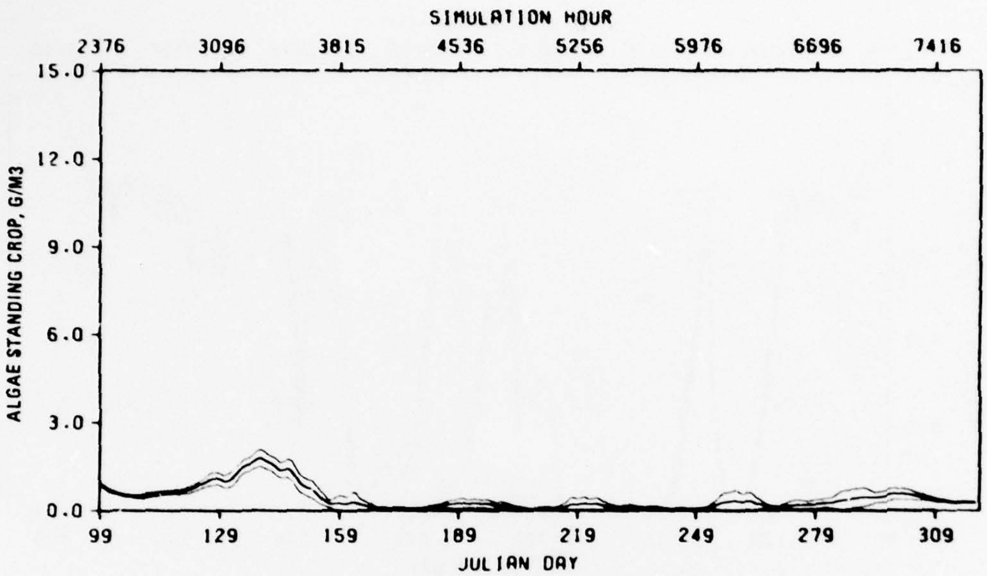


Figure 58. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, bottom withdrawal, 1971

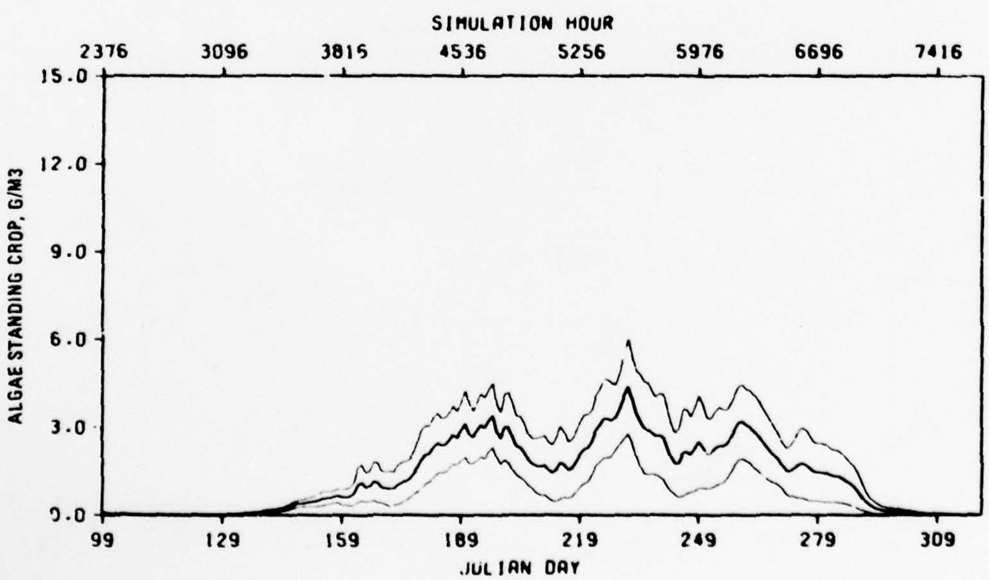


Figure 59. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, bottom withdrawal, 1971

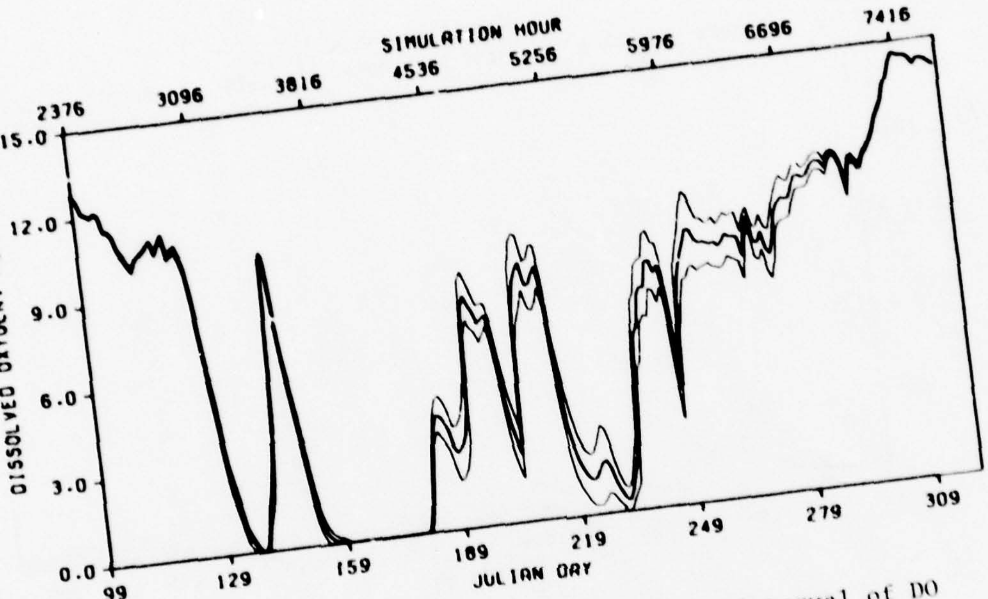


Figure 60. Mean and 95 percent confidence interval of DO in the bottom metre, surface withdrawal, 1971

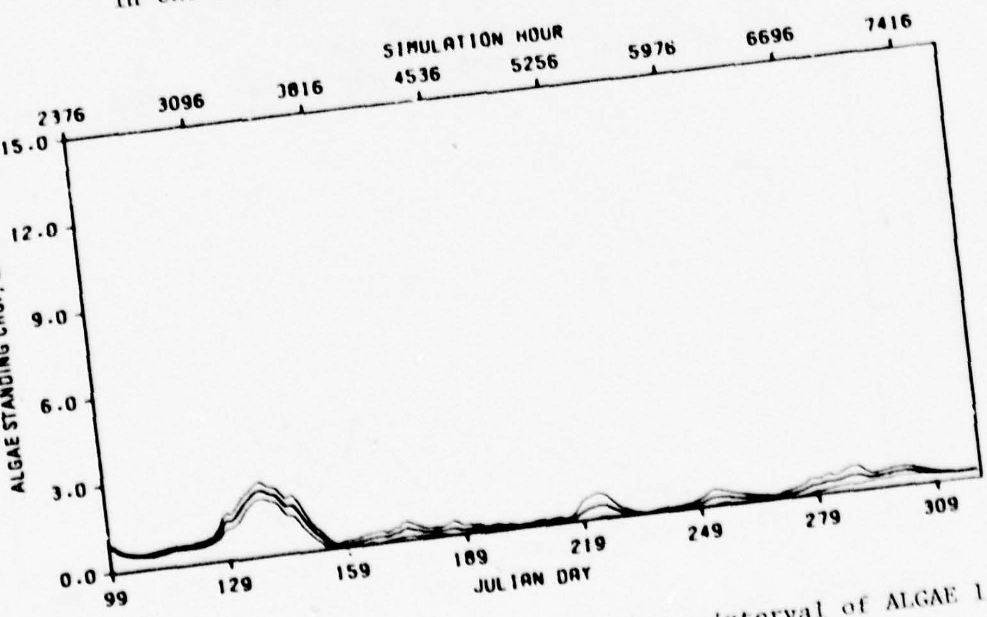


Figure 61. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, surface withdrawal, 1971

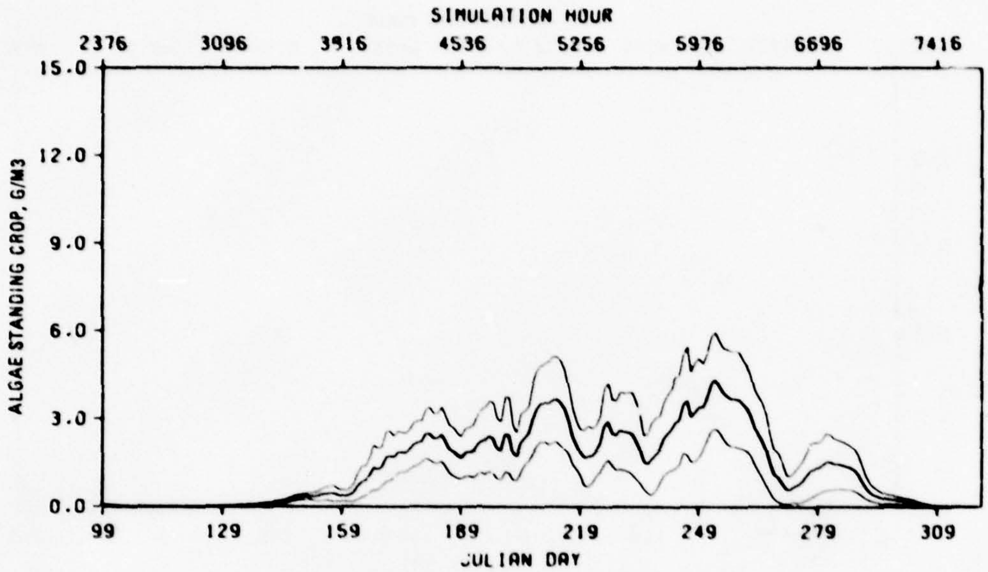


Figure 62. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, surface withdrawal, 1971

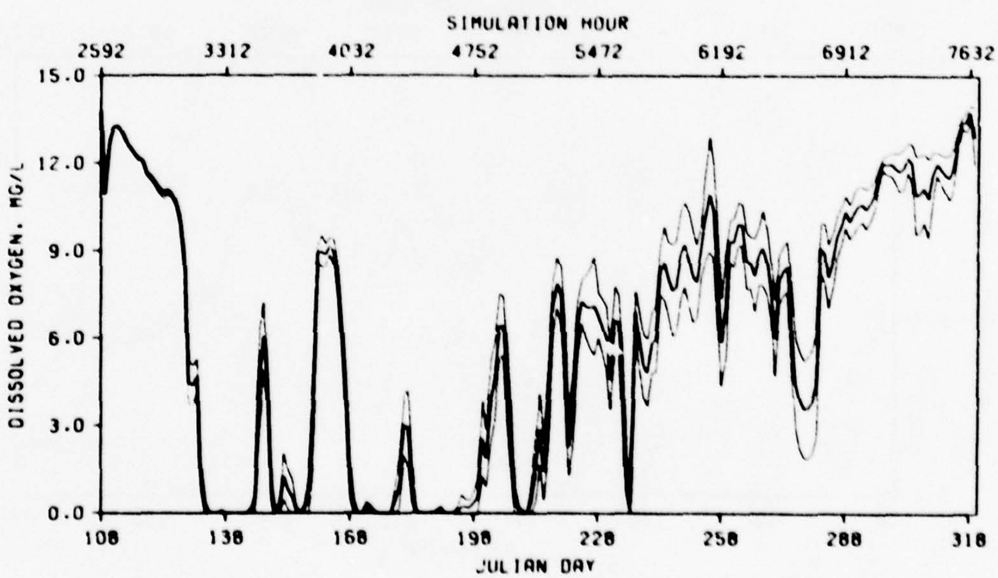


Figure 63. Mean and 95 percent confidence interval of DO in the bottom metre, bottom withdrawal, 1975

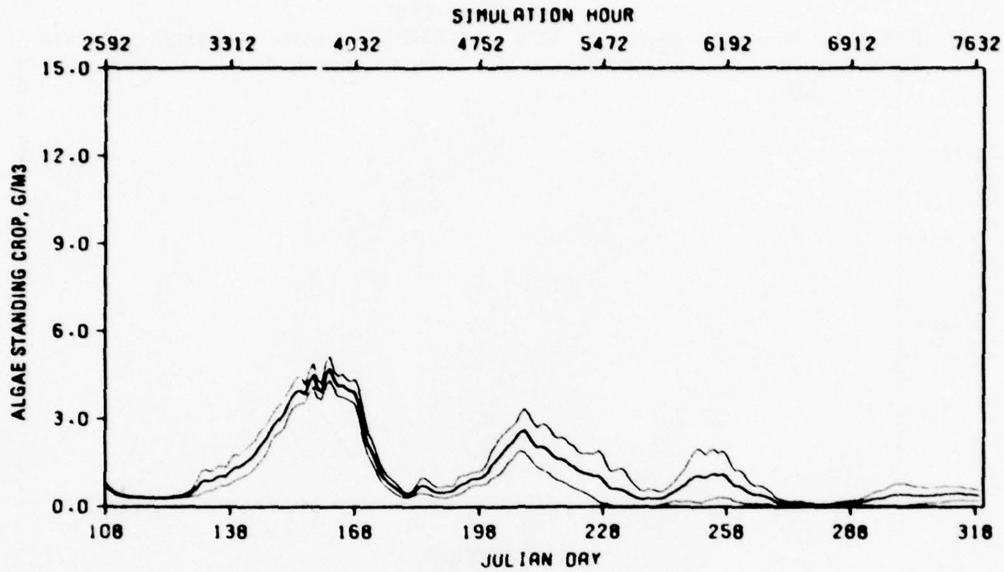


Figure 64. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, bottom withdrawal, 1975

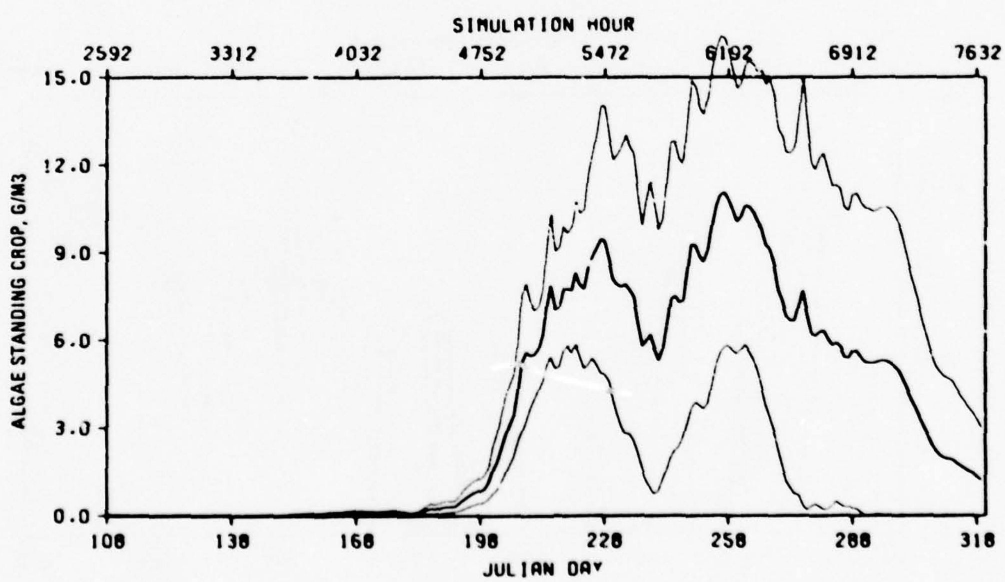


Figure 65. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, bottom withdrawal, 1975

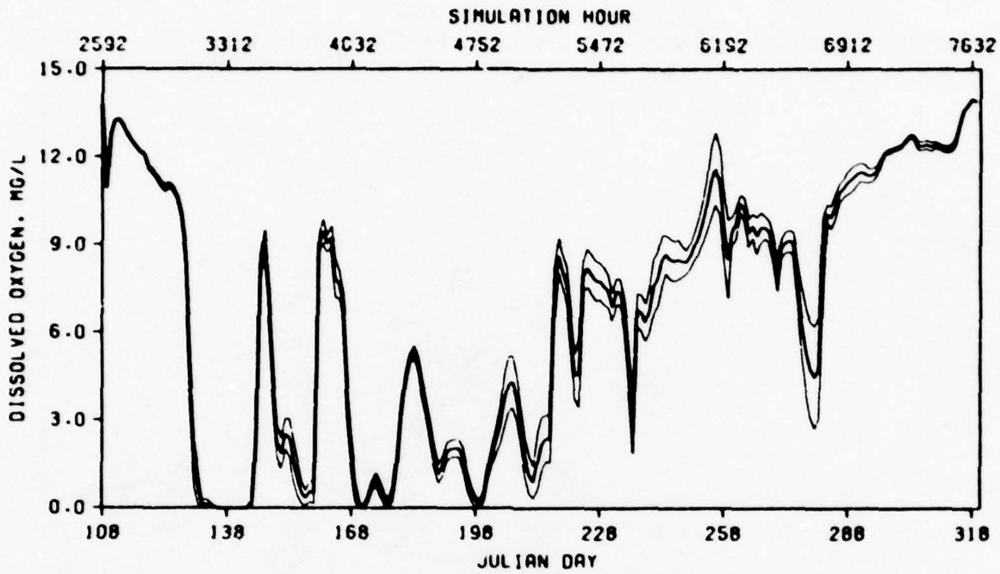


Figure 66. Mean and 95 percent confidence interval of DO in the bottom metre, surface withdrawal, 1975

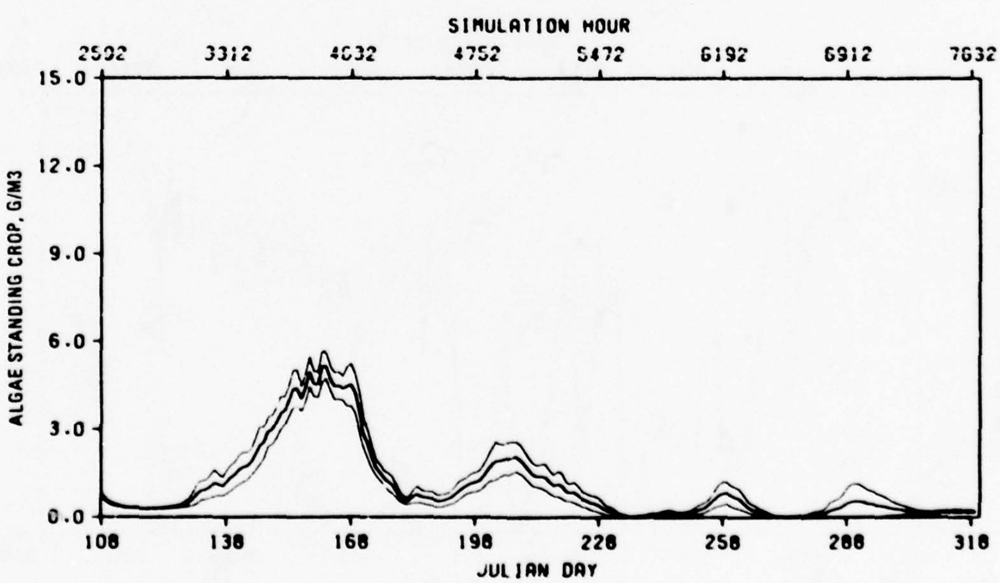


Figure 67. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, surface withdrawal, 1975

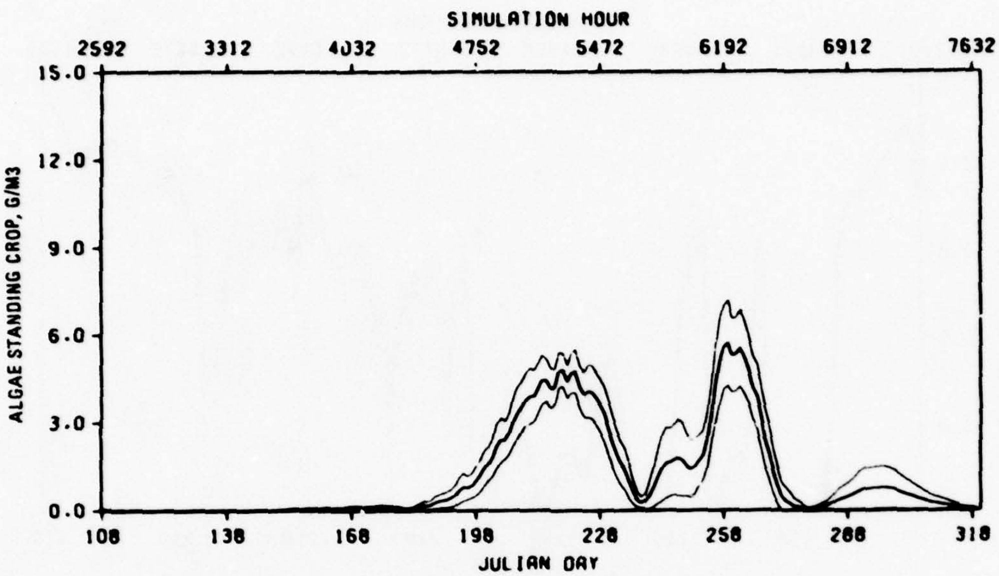


Figure 68. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, surface withdrawal, 1975

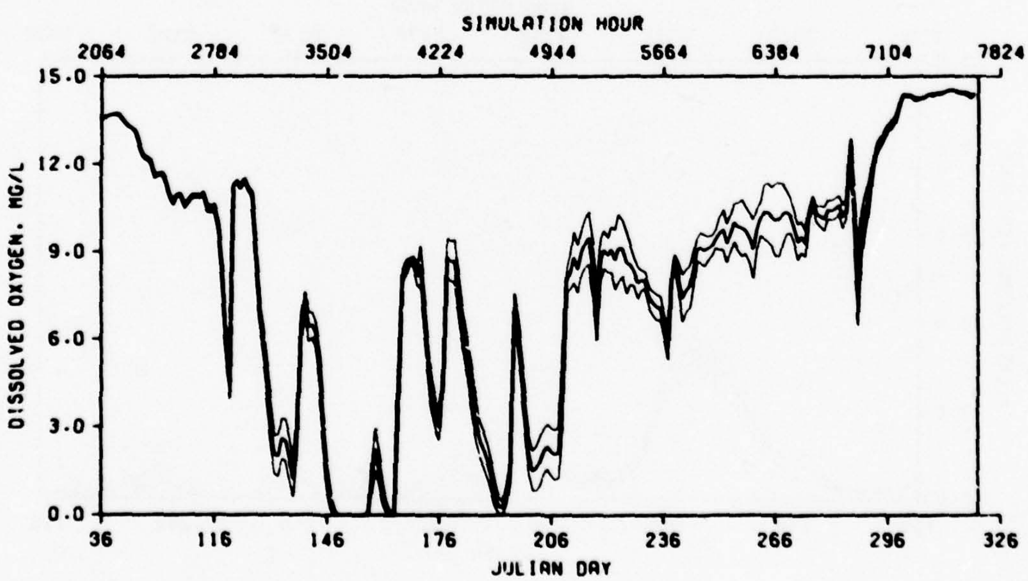


Figure 69. Mean and 95 percent confidence interval of DO in the bottom metre, bottom withdrawal, 1976

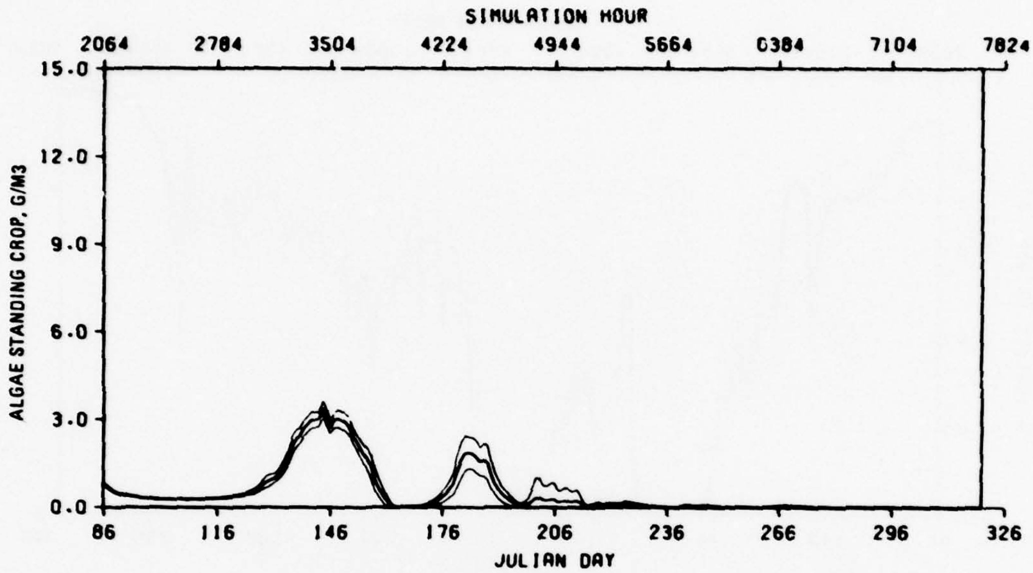


Figure 70. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, bottom withdrawal, 1976

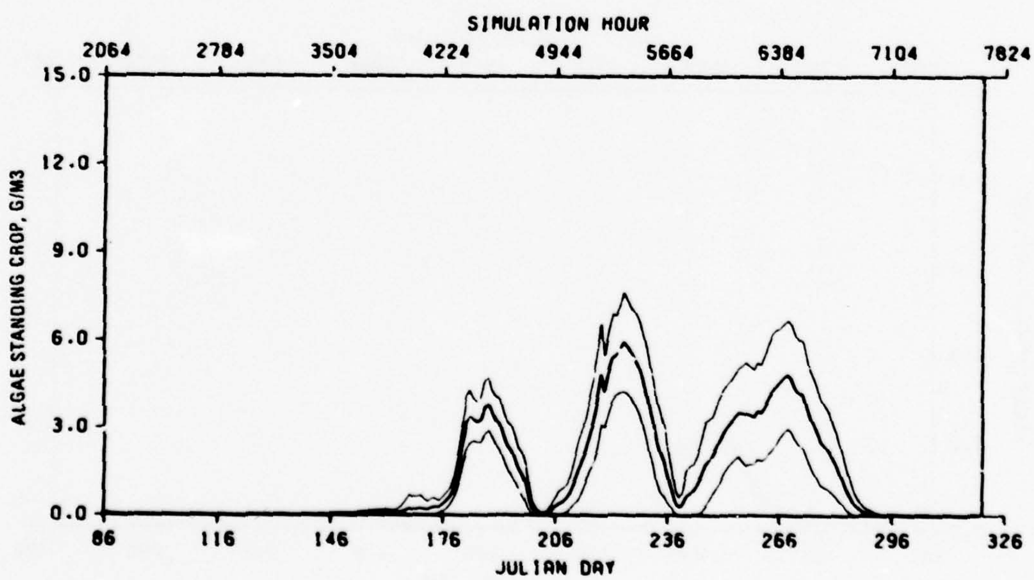


Figure 71. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, bottom withdrawal, 1976

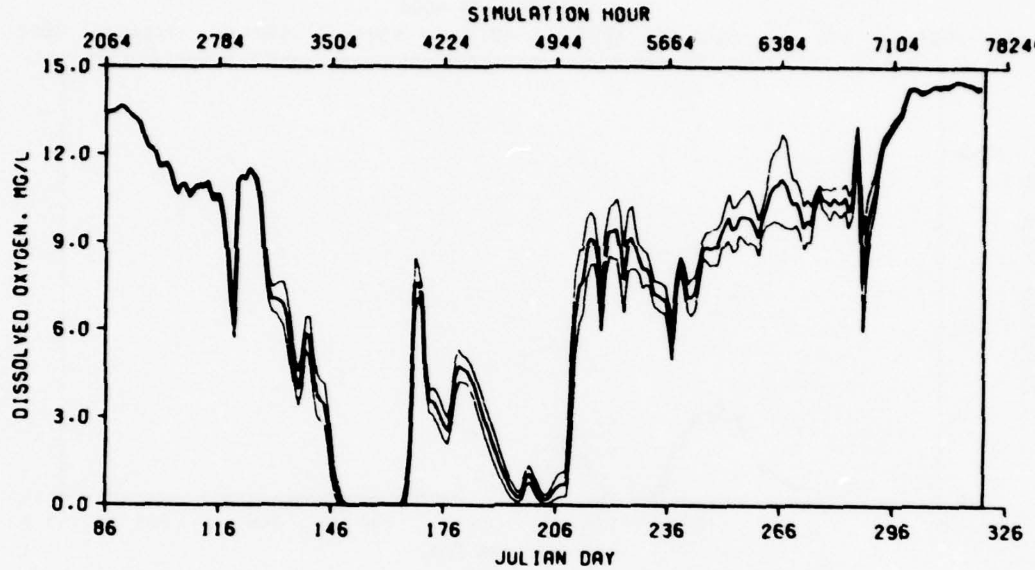


Figure 72. Mean and 95 percent confidence interval of DO in the bottom metre, surface withdrawal, 1976

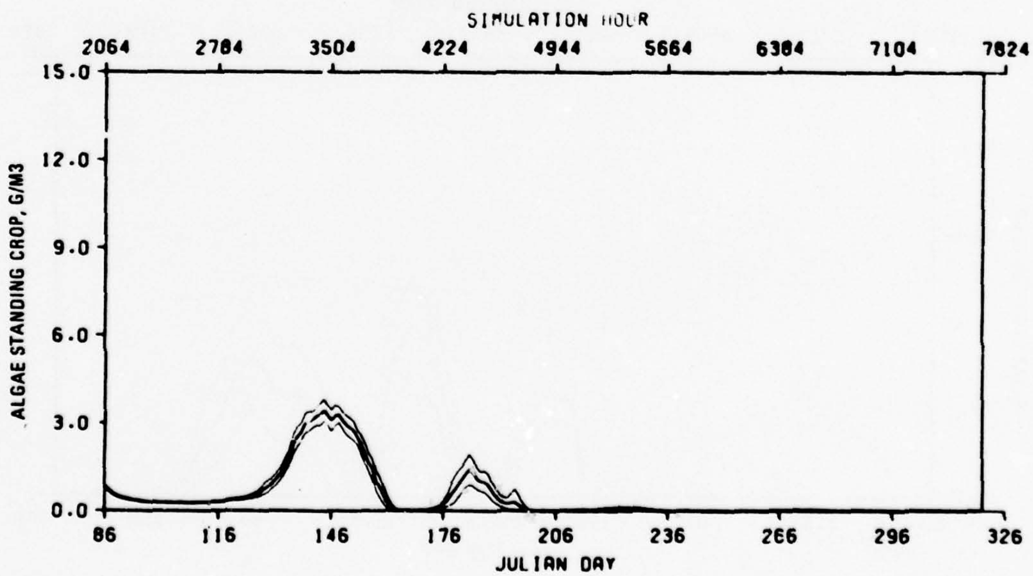


Figure 73. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, surface withdrawal, 1976

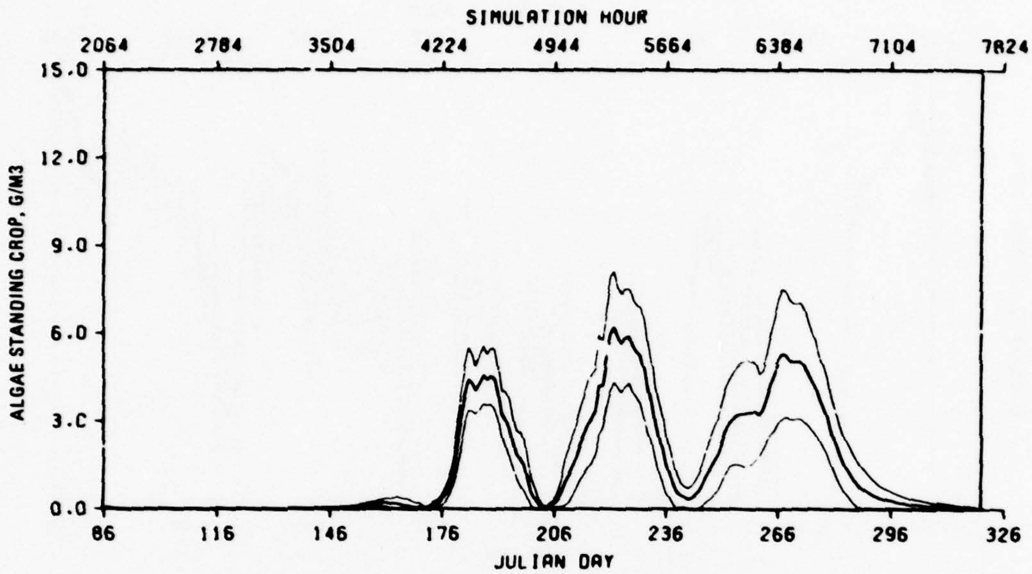


Figure 74. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, surface withdrawal, 1976

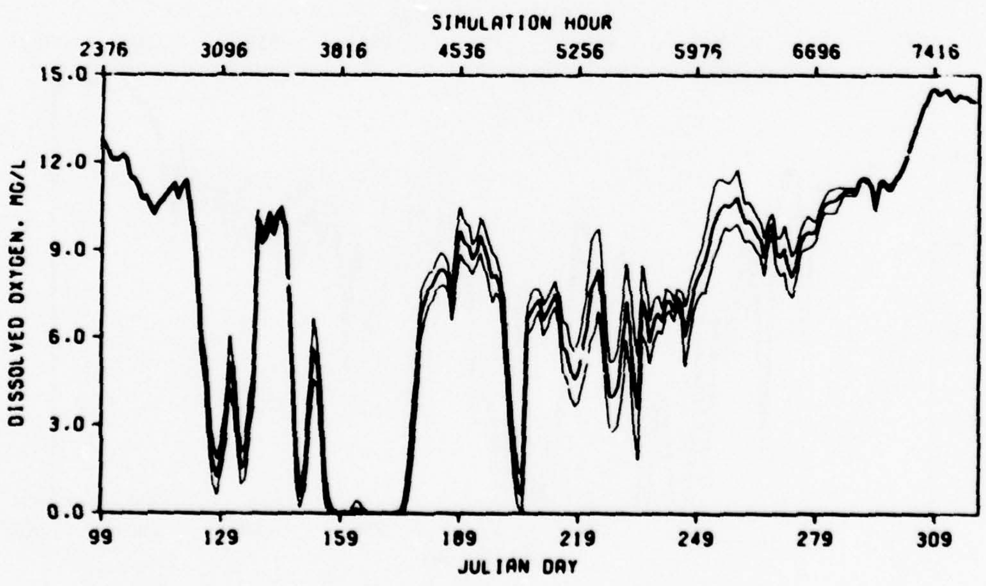


Figure 75. Mean and 95 percent confidence limits of DO in the bottom metre, pool el 322.5 m msl, 1971

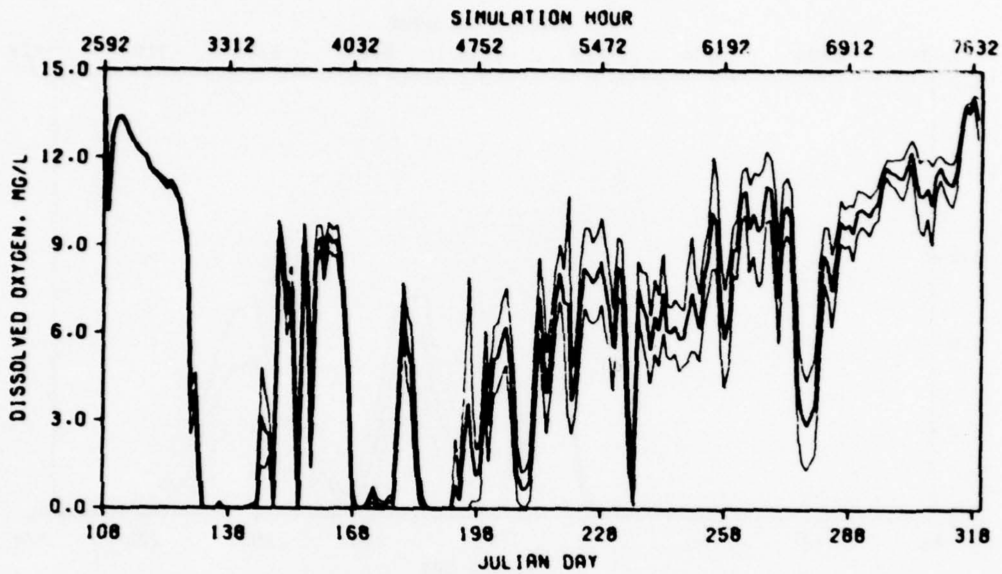


Figure 76. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 322.5 m msl, 1975

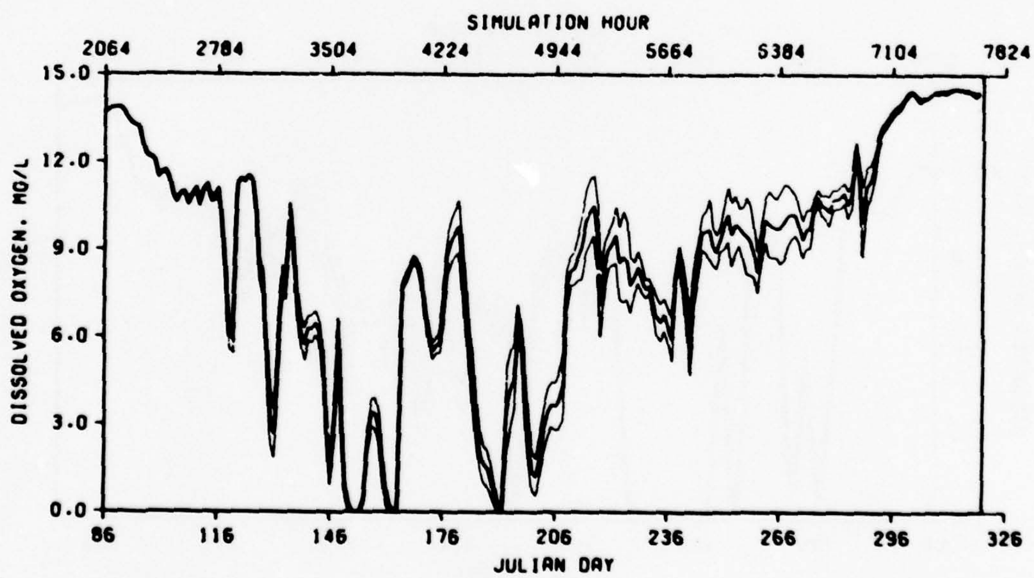


Figure 77. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 322.5 m msl, 1976

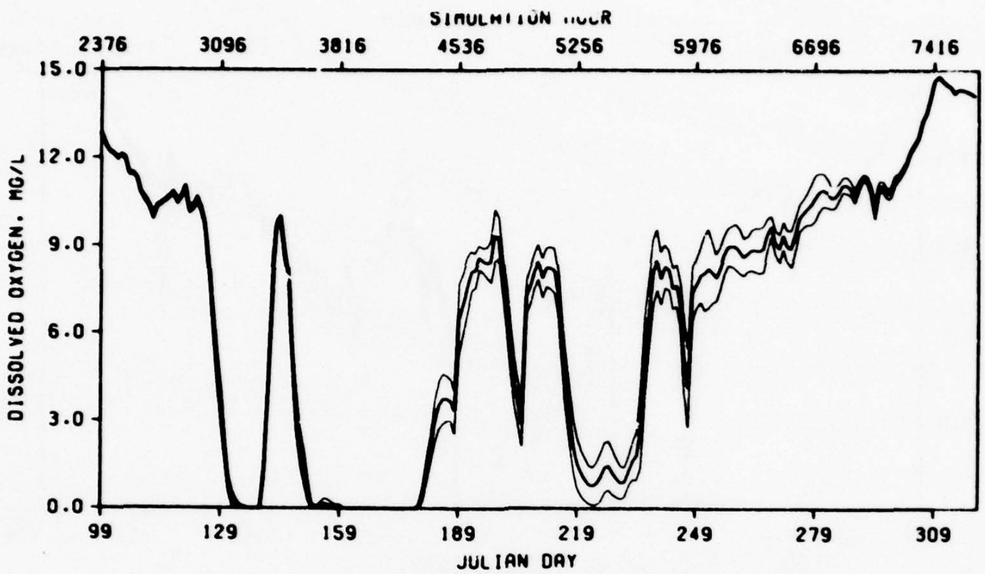


Figure 78. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 325.5 m msl, 1971

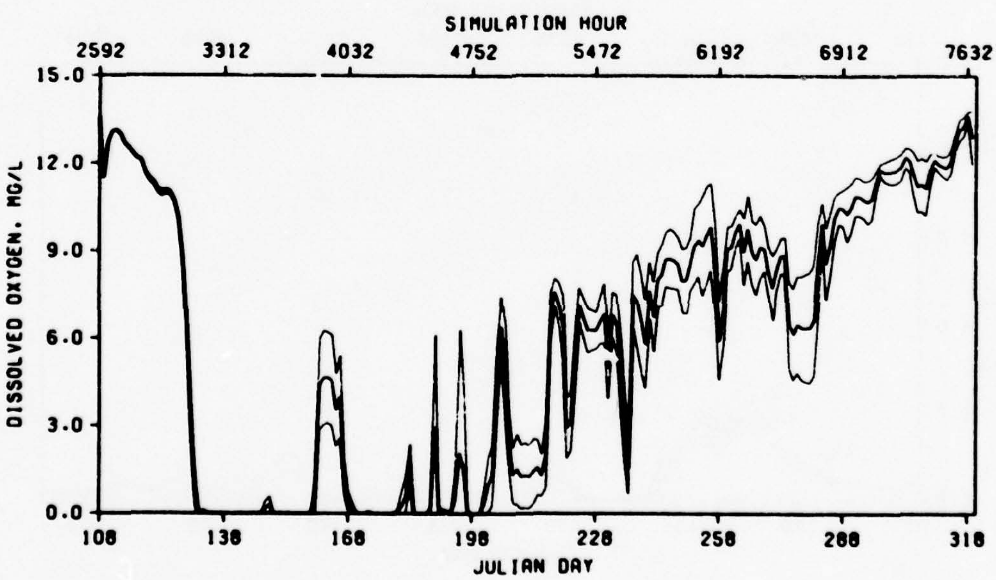


Figure 79. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 325.5 m msl, 1975

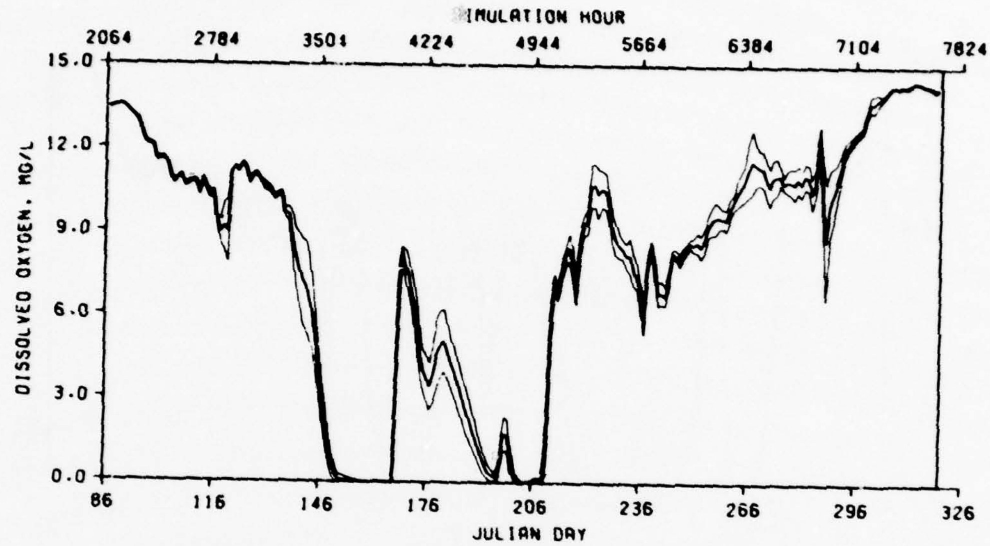


Figure 80. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 325.5 m msl, 1976

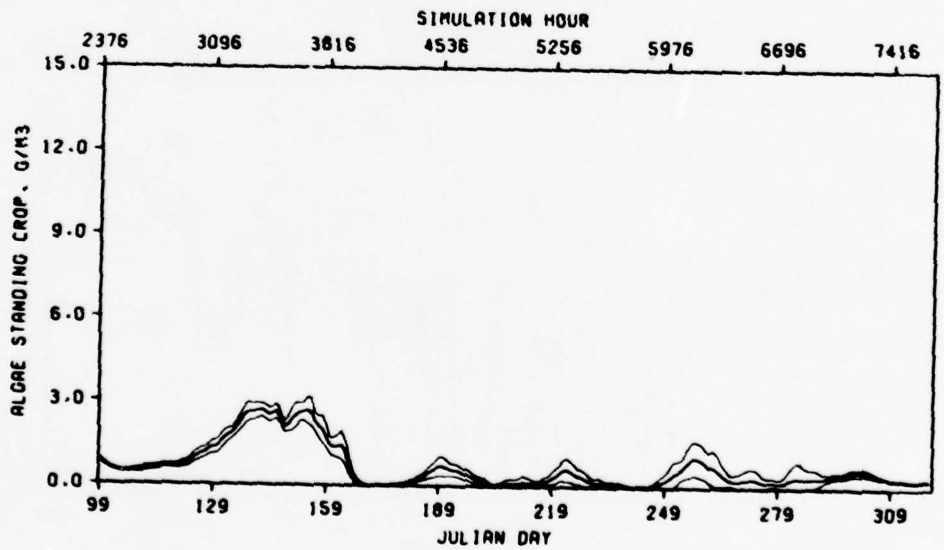


Figure 81. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 322.5 m msl, 1971

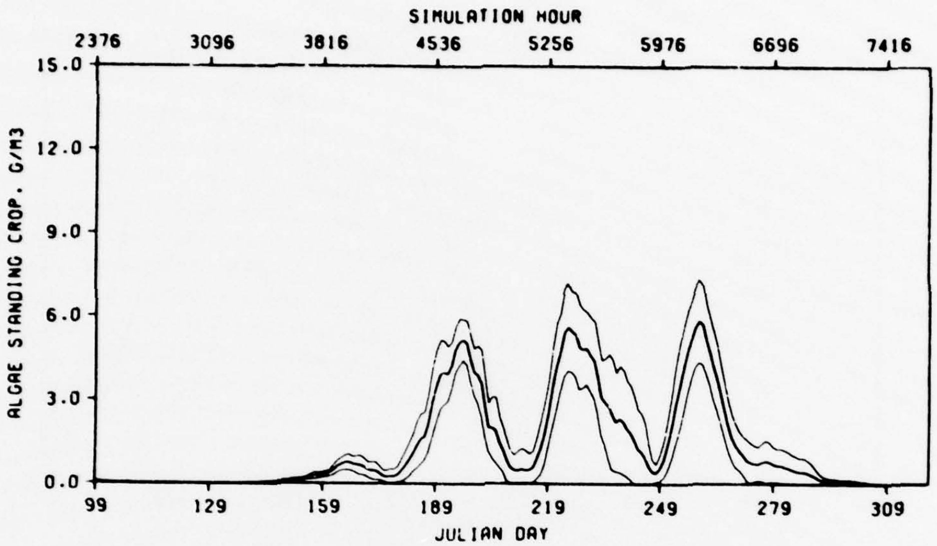


Figure 82. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 322.5 m msl, 1971

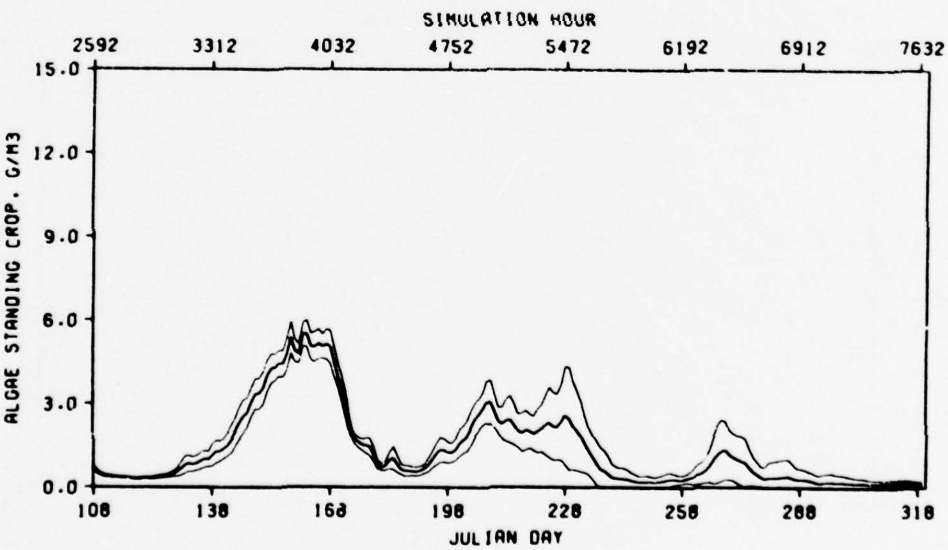


Figure 83. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 322.5 m msl, 1975

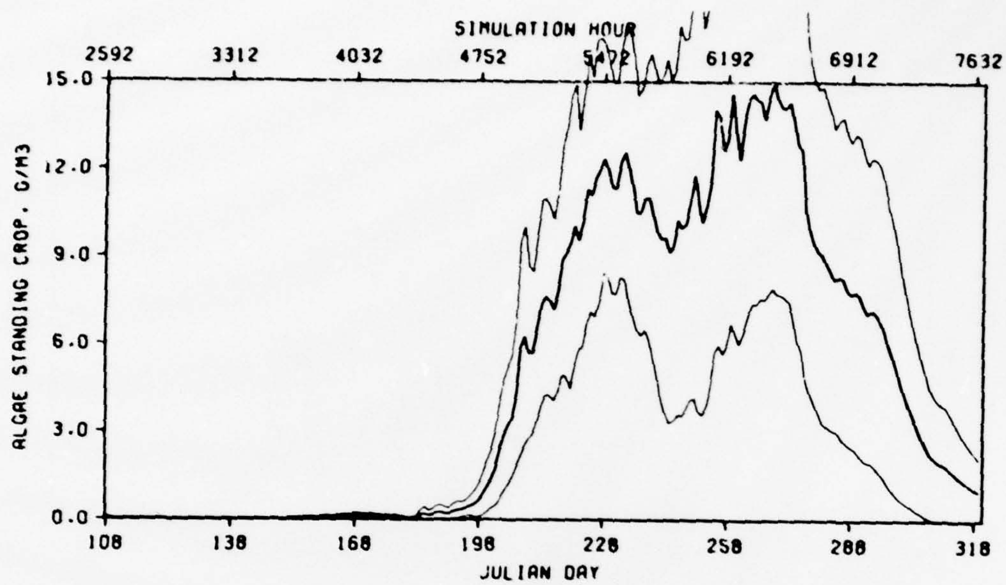


Figure 84. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 322.5 m msl, 1975

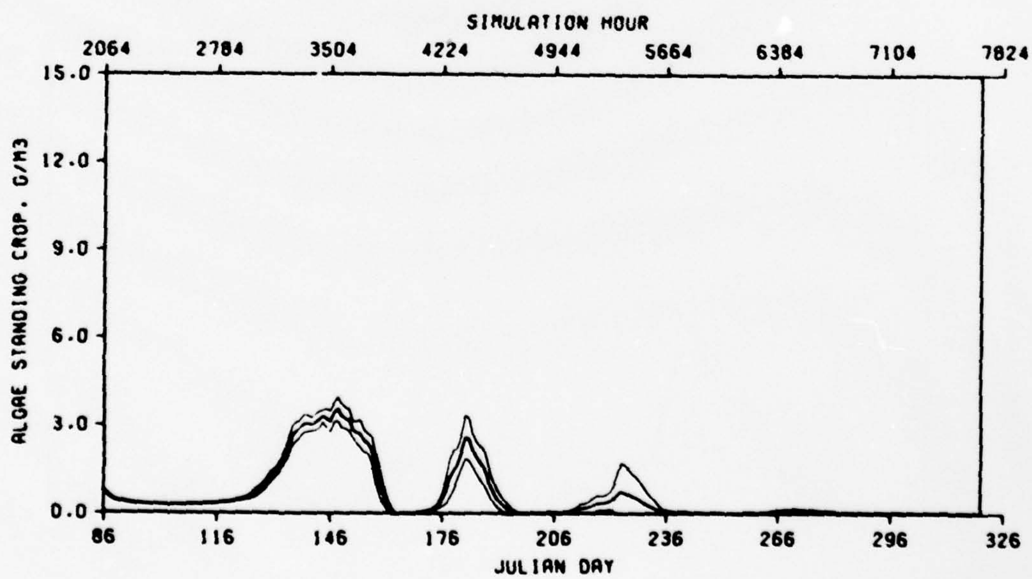


Figure 85. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 322.5 m msl, 1976

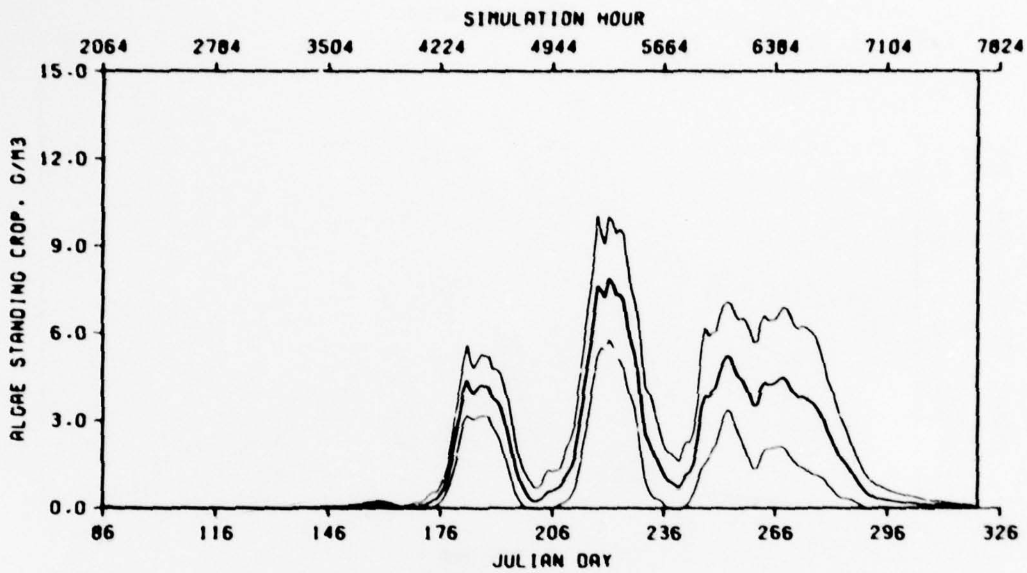


Figure 86. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 322.5 m msl, 1976

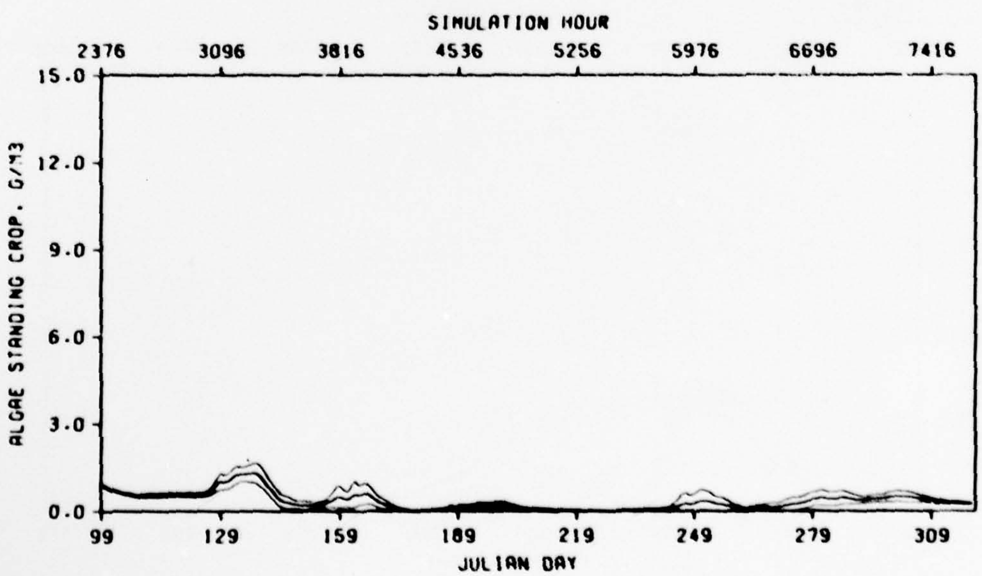


Figure 87. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 325.5 m msl, 1971

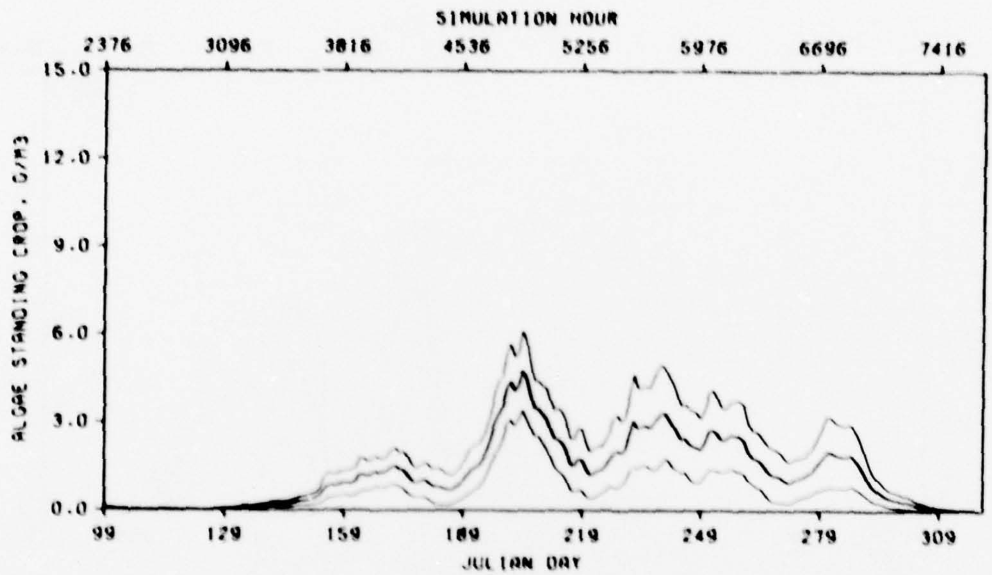


Figure 88. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 325.5 m msl, 1971

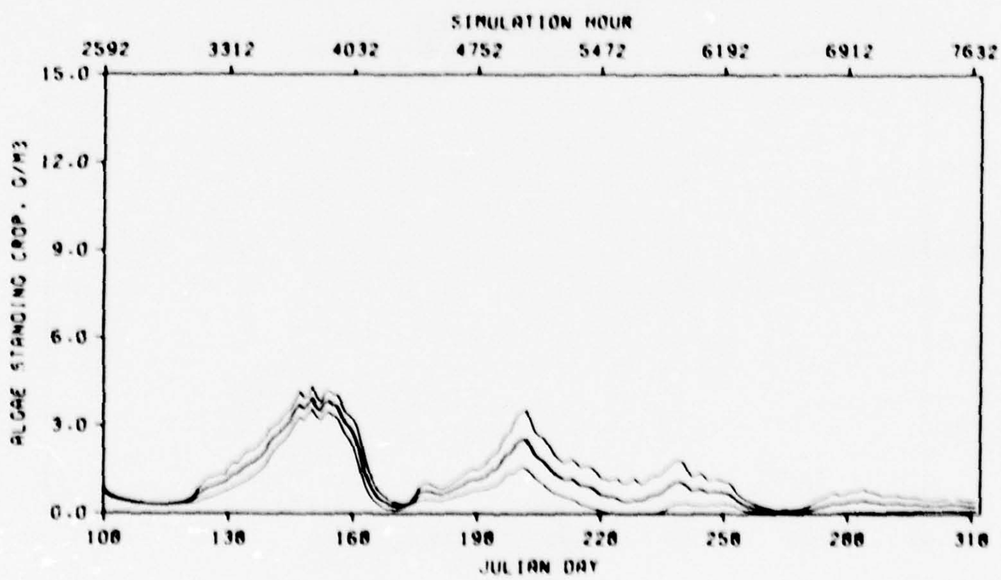


Figure 89. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 325.4 m msl, 1975

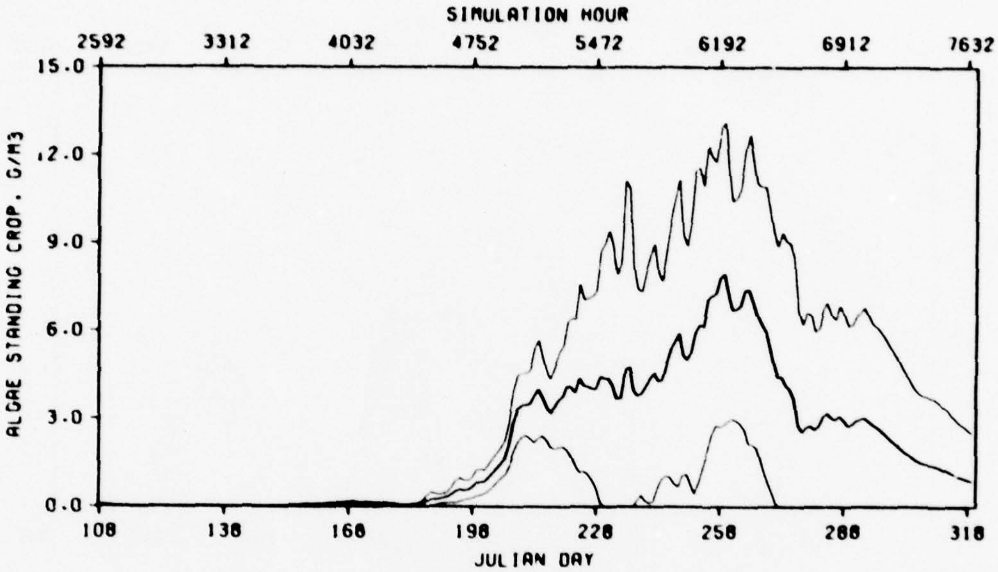


Figure 90. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 325.5 m msl, 1975

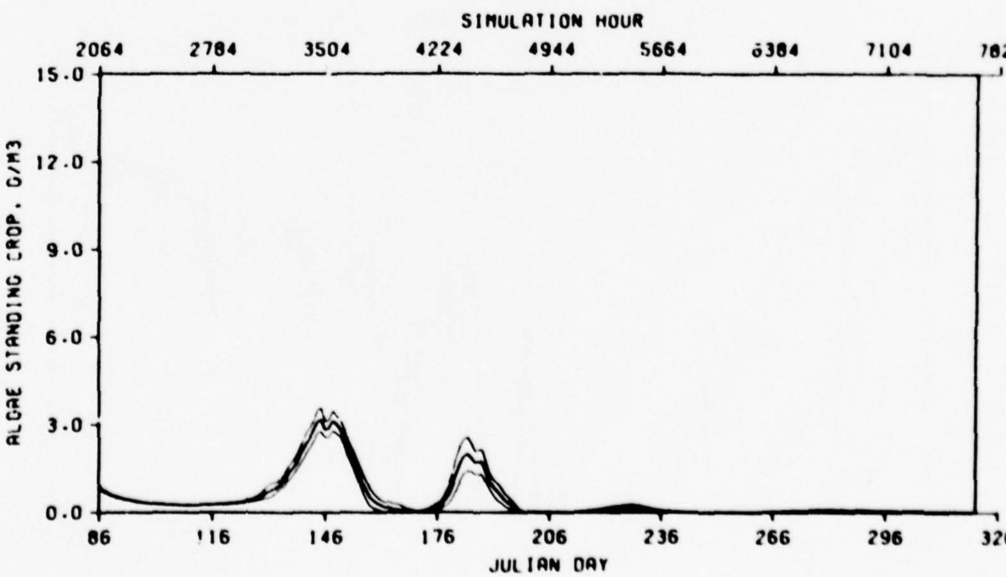


Figure 91. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 325.5 m msl, 1976

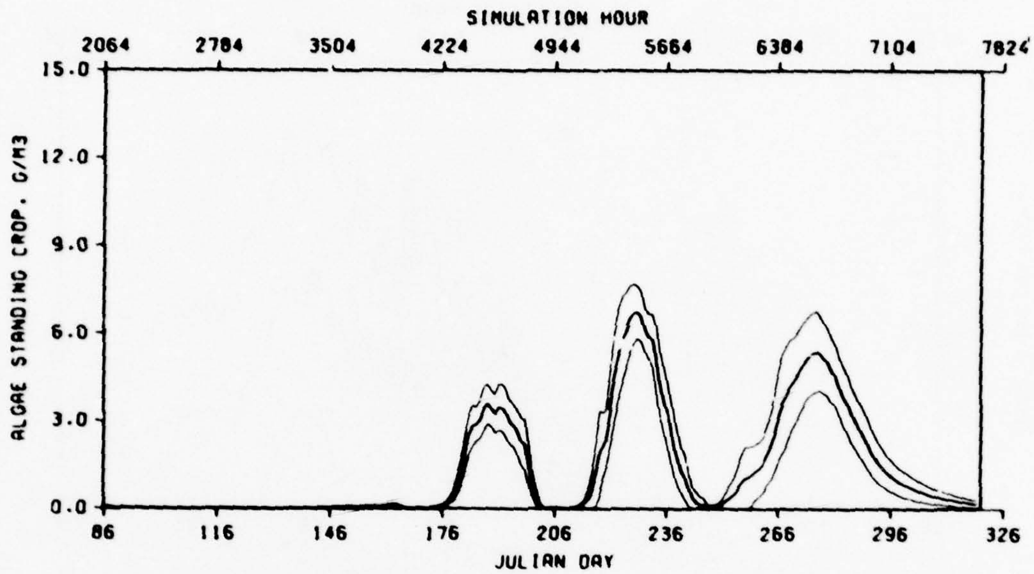


Figure 92. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 325.5 m msl, 1976

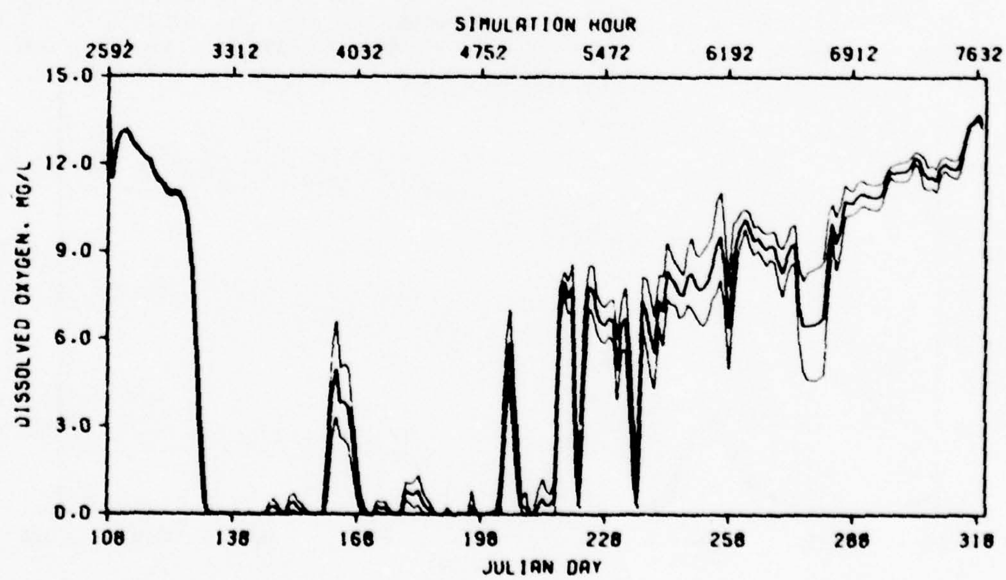


Figure 93. Mean and 95 percent confidence interval of DO in the bottom metre, pool el 325.5 m msl, selective withdrawal, 1975

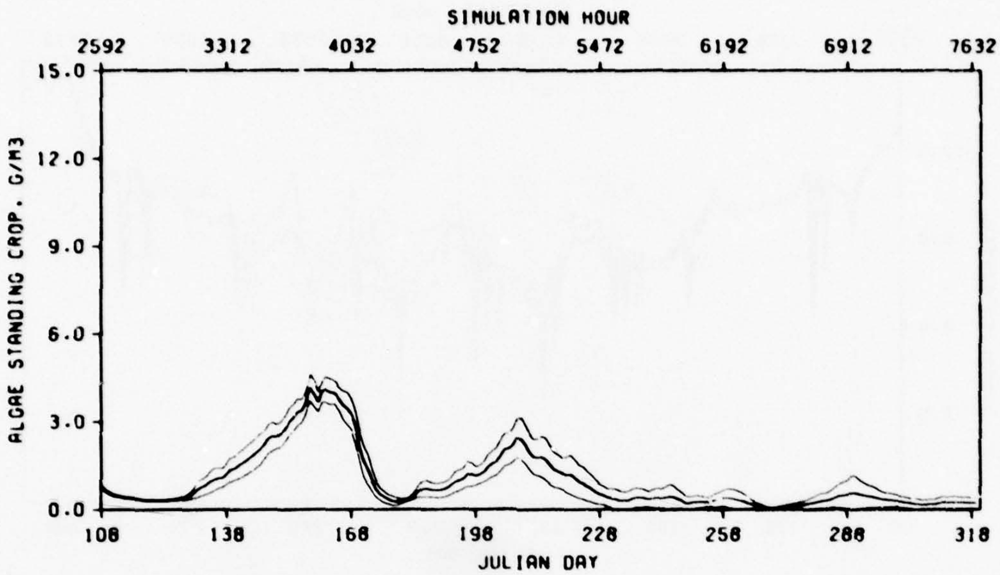


Figure 94. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, pool el 325.5 m msl, selective withdrawal, 1975

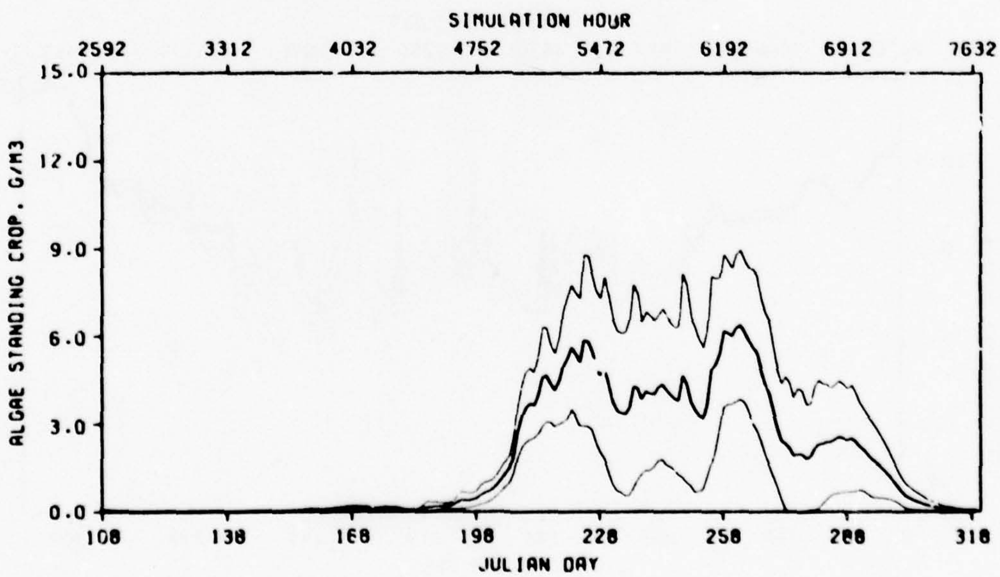


Figure 95. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, pool el 325.5 m msl, selective withdrawal, 1975

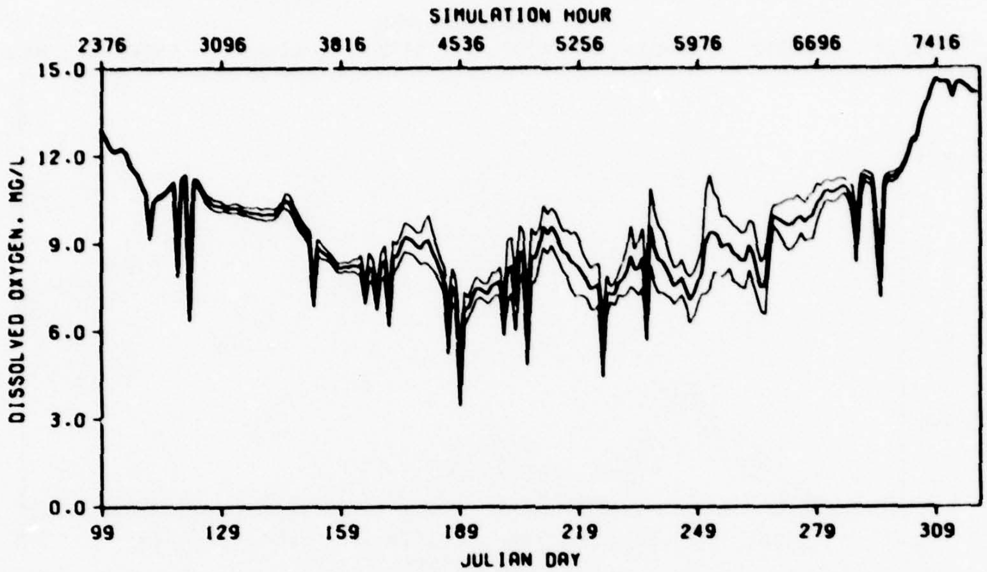


Figure 96. Mean and 95 percent confidence interval of DO in the bottom metre, assuming destratification, 1971

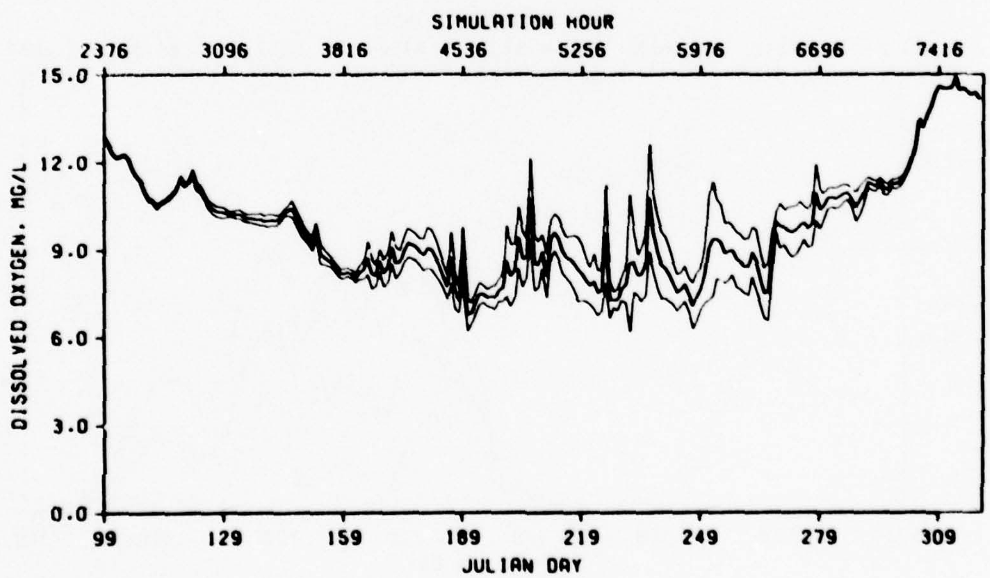


Figure 97. Mean and 95 percent confidence interval of DO in the surface waters, assuming destratification, 1971

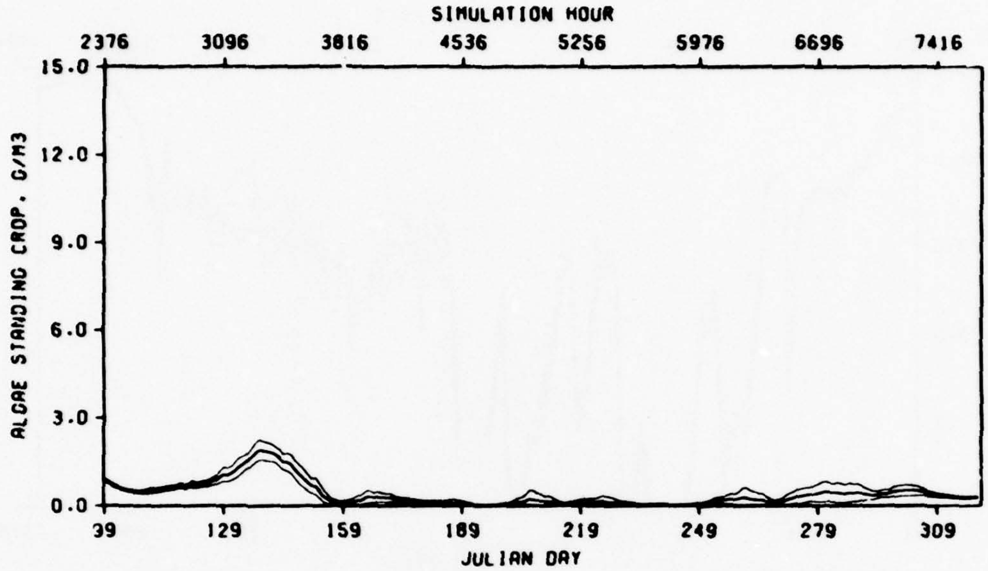


Figure 98. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone, assuming destratification, 1971

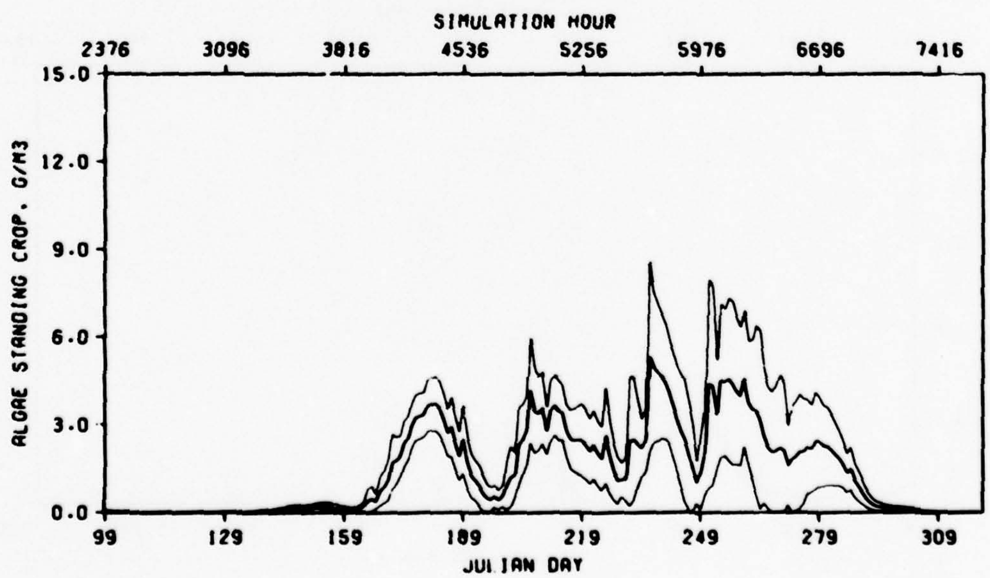


Figure 99. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone, assuming destratification, 1971

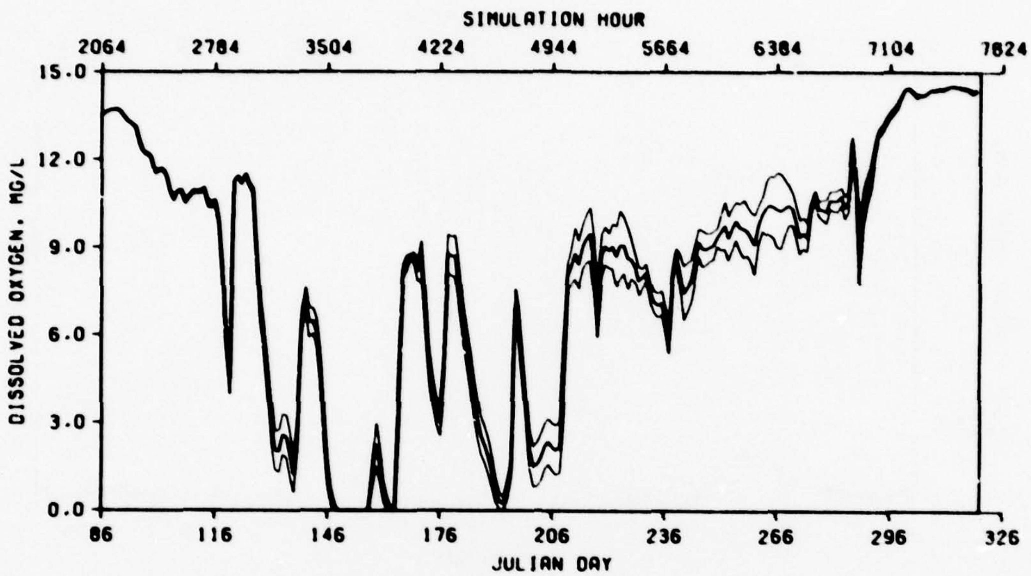


Figure 100. Mean and 95 percent confidence interval of DO in the bottom metre with a minimum of release of 0.42 m^3/sec, 1976

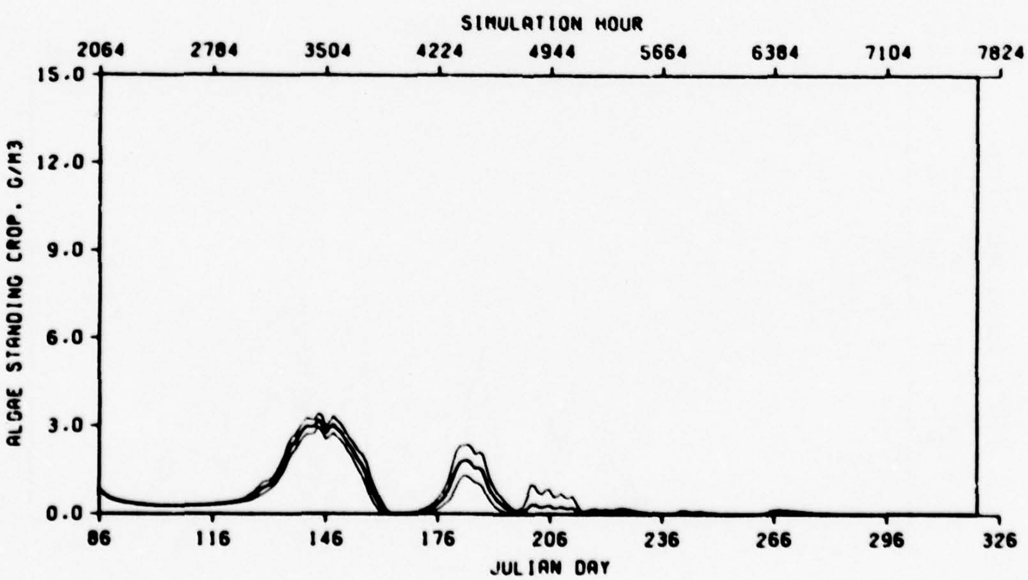


Figure 101. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone with a minimum release of 0.42 m^3/sec, 1976

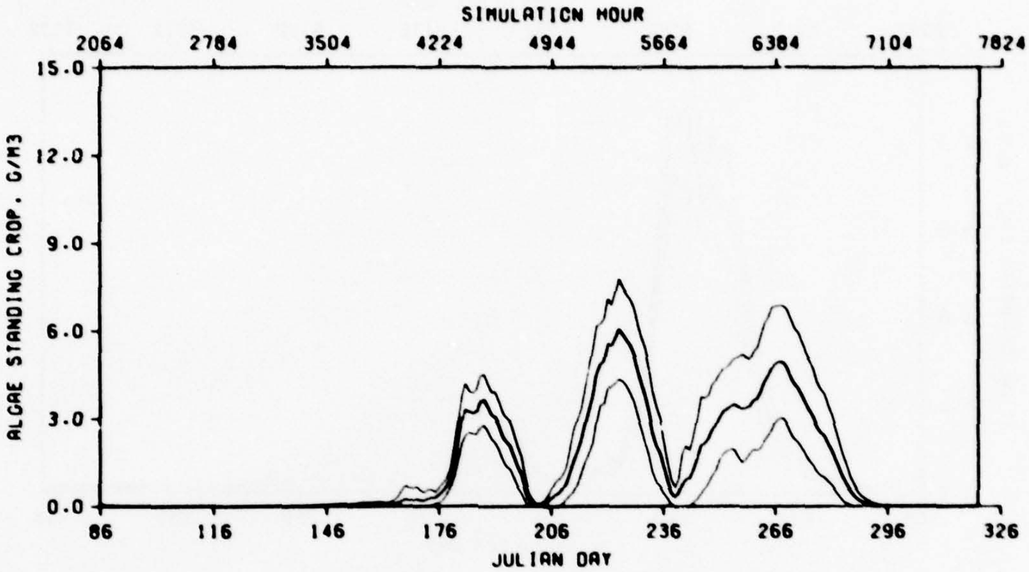


Figure 102. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone with a minimum release of 0.42 m^3/sec, 1976

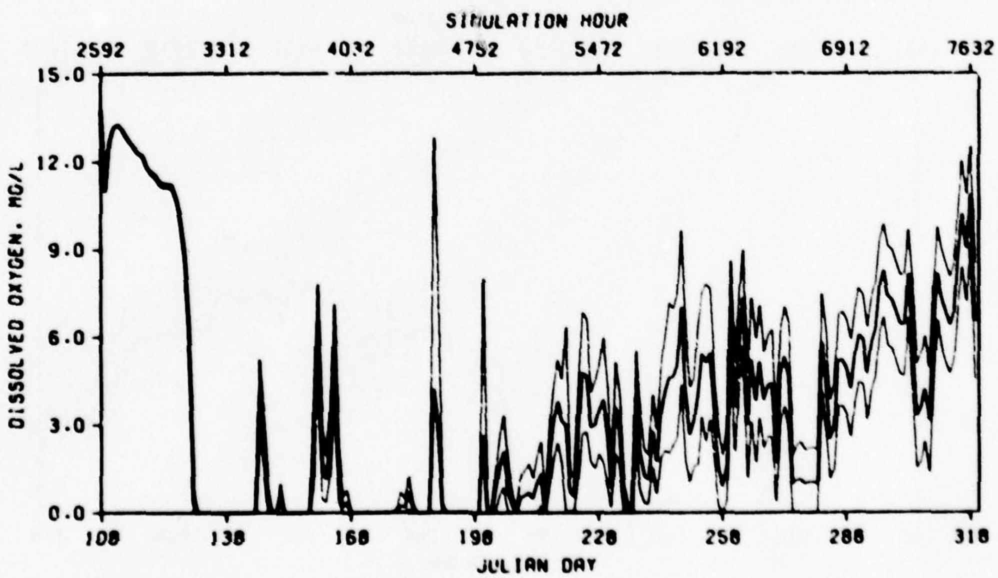


Figure 103. Mean and 95 percent confidence interval of DO in the bottom metre with a maximum release of 48 m^3/sec, 1975

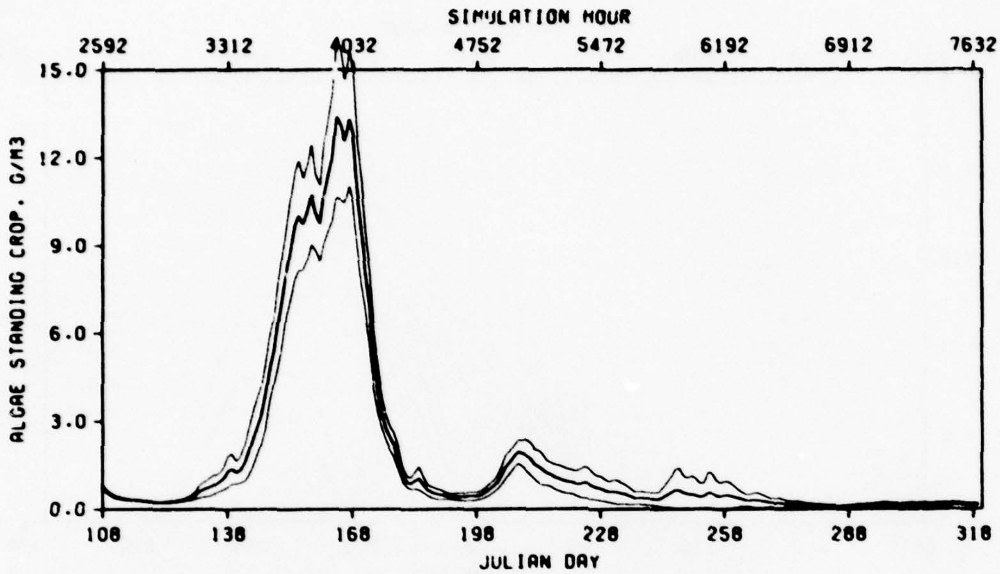


Figure 104. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone with a maximum release of 48 m^3/sec, 1975

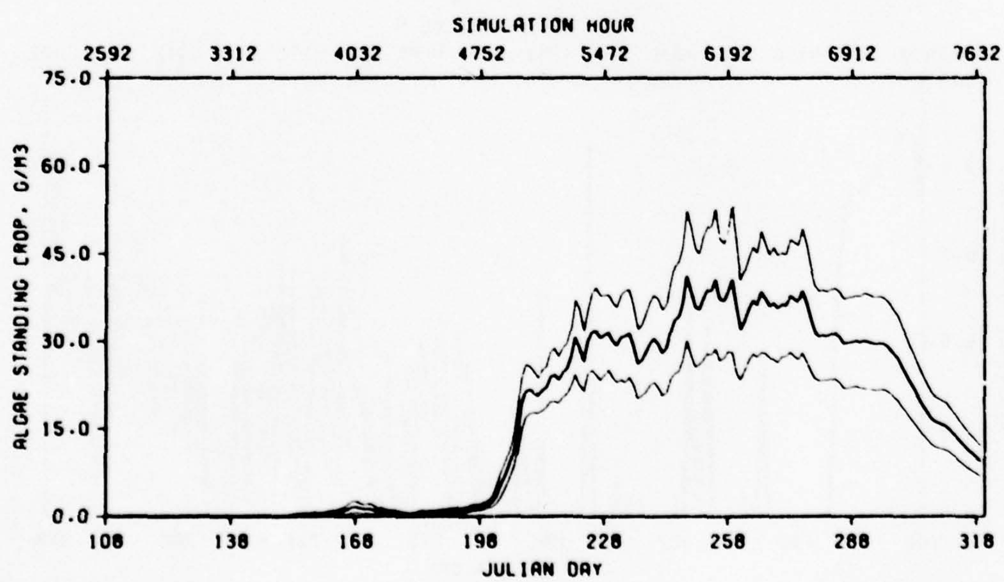


Figure 105. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone with a maximum release of 48 m^3/sec, 1975

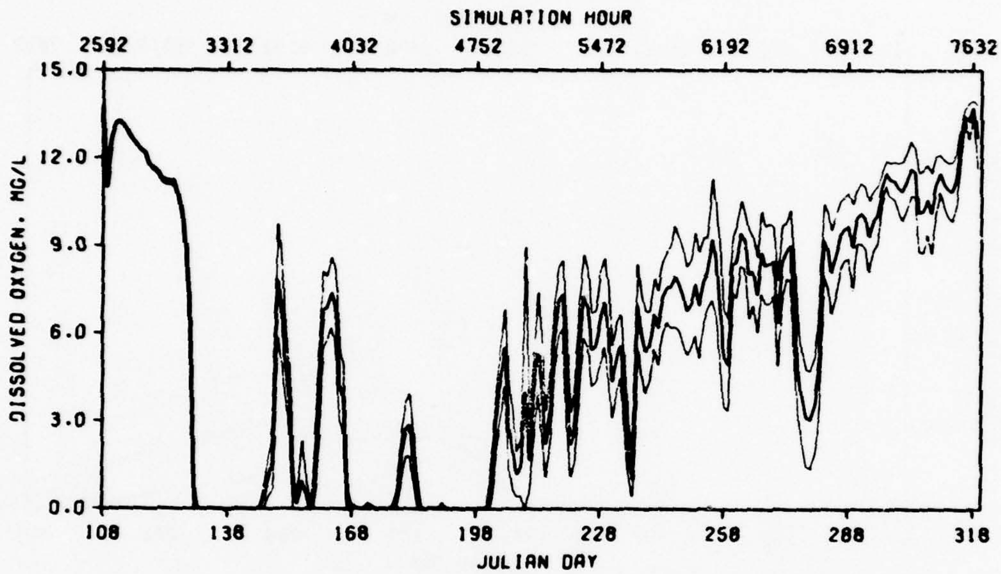


Figure 106. Mean and 95 percent confidence interval of DO in the bottom metre with a maximum release of 48 m^3/sec, selective withdrawal. 1975

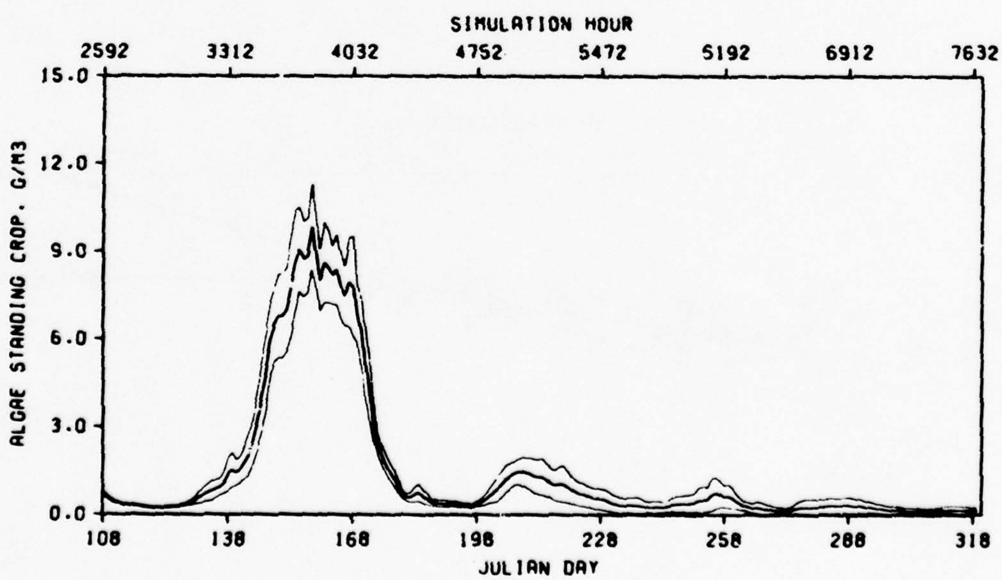


Figure 107. Mean and 95 percent confidence interval of ALGAE 1 in the euphotic zone with a maximum release of 48 m^3/sec, selective withdrawal, 1975

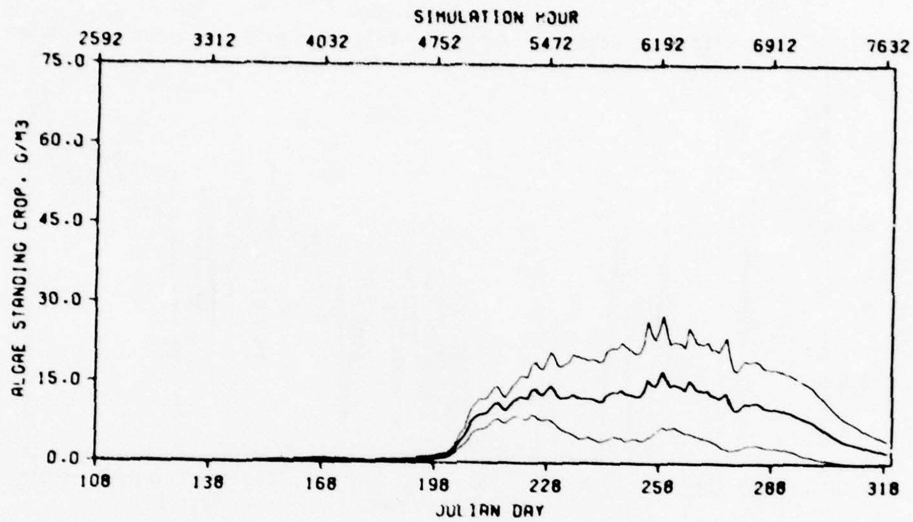


Figure 108. Mean and 95 percent confidence interval of ALGAE 2 in the euphotic zone with a maximum release of 48 m^3/sec, selective withdrawal, 1975

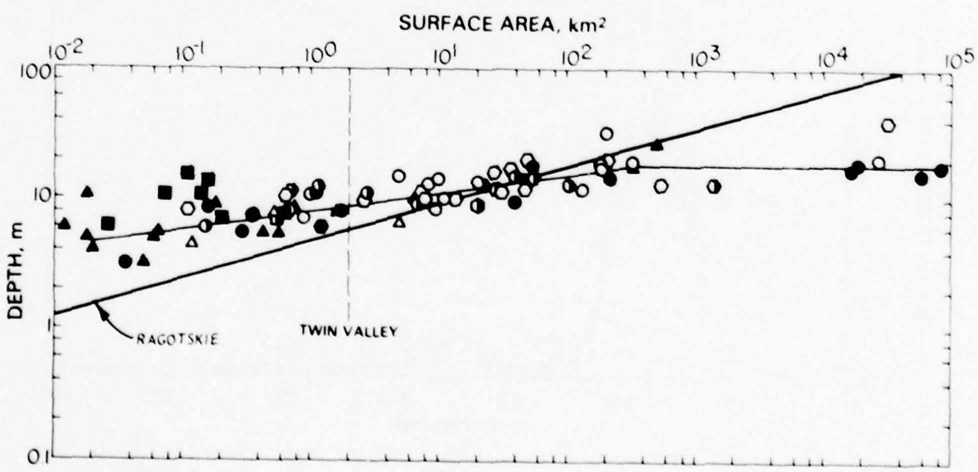


Figure 109. Thermocline depth as a function of surface area

APPENDIX A:

REPORT ON ALGAL ASSAY PROCEDURES: BOTTLE TEST BIOASSAYS OF WATER SAMPLES

by

Leslie A. Gardner

Utah Water Research Laboratory*
Utah State University
Logan, Utah

August 1978

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* UWRL is an EPA-certified laboratory through the Utah State Division of Health, Salt Lake City.

LIMITING NUTRIENT BIOASSAYS

RUN 1 (APRIL 1978)

Water samples arrived at the Utah Water Research Laboratory via air freight from Fargo, North Dakota, on April 12 and 13, 1978. The samples were shipped in polyethylene containers and arrived on partial ice. The samples were designated the following numbers and will be referred to as such throughout this report.

1. Wild Rice River at Twin Valley Gauge
   10 April 1978
2. Inflow to Lower Rice Lake
   10 April 1978
3. Wild Rice River Upper End Conservation Pool
   10 April 1978
4. Dayton Hollow Dam Mid Pool, Depth Integrated
   11 April 1978
5. Ottertail River Inlet to Dayton Hollow
   11 April 1978

Sample Pretreatment

Immediately on arrival approximately 3 ℓ of each sample was filter sterilized using 0.45-μ Millipore membrane filters. Filtering removes native algae from the test water and enables the use of unialgal test species in the bioassay.

Upon completion of filtering the samples were subjected to routine chemical analyses for the determination of indigenous levels of soluble phosphorus and soluble inorganic nitrogen (Table A1).

Prior to use in the bioassays all glass and labware contacting algae were cleaned in the following manner: sodium bicarbonate wash, tap water rinse, 1:1 hydrochloric acid rinse, deionized water rinse, and, finally, ultra pure deionized water rinse. Following the washing procedure all glassware was autoclaved using an aluminum foil closure at 121°C for 15 minutes.

Experimental Setup Procedure

The bioassays were conducted using 100-ml sample volumes in 500-ml Erlenmeyer flasks. Inverted beakers were chosen for flask closures in order to permit good CO_2-O_2 exchange and prevent contamination.

Each of the five test waters received the following treatments. All treatments for all samples were set up in triplicate. The sample blank (Treatment A) was included to provide the basis for comparison of the other treatments and provide a measure of the general fertility of the sample.

Treatment

A. Sample

B. Sample + 2.1 mg/ℓ NO_3-N

C. Sample + 0.093 mg/ℓ PO_4-P

D. Sample + 2.1 mg/ℓ NO_3-N + 1.093 mg/ℓ PO_4-P

E. Sample + AAM* levels of trace elements

F. Sample + AAM levels of HCO_3^-

G. Sample + 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of: trace elements, HCO_3, $CaCl_2$, and $MgSO_4$

Control. Distilled water + 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of trace elements, HCO_3^-, $CaCl_2$, and $MgSO_4$

Constituents of AAM are listed on Table A2. The samples and control contained one-half AAM levels of nitrogen and phosphorus whereas all other constituents were added at full strength levels.

The control treatment was included to provide a general check on cell growth and to provide an index for comparing growth levels in the test waters. The results in Figures A1-A10 are all compared to the AAM control.

Algal bioassays were performed according to EPA (1971) using the green alga, *Selenastrum capricornutum* PRINTZ. The test flasks were

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* AAM = algal assay medium.

placed in a constant temperature room ($24^oC \pm 2^oC$) with "cool-white" fluorescent lighting providing illumination of 400 fc (4304 lx) ± 10 percent.

The assays were monitored by determining the optical density (OD, Bausch and Lomb Spec 70, 750 nm, 1-cm path length) and relative fluorescence (RF × 30, Turner Fluorometer, Model 110). Optical density was measured over a 12-day period while relative fluorescence was measured to monitor the progress of the cultures for the first 6 to 7 days when optical density does not provide a great deal of sensitivity. The results of both determinations are represented graphically in Figures A1-A10. Maximum values for optical density are listed on Table A3.

Optical density is an indirect means of measuring algal cell biomass. As a consequence, OD is linearly related to biomass as dry weight (Porcella et al. 1973). Due to this linearity, biomass, as volatile suspended solids (VSS), can be calculated directly from OD. The relationship used to convert OD to VSS in Table A4 is:

$$\text{VSS, mg}/\ell = 350(\text{OD}) + 3.5 \qquad \text{(A1)}$$

It is the algae yield as dry weight which will be used to substantiate the limiting nutrient in each of the test waters. Figures A11-A20 represent the maximum growth, as VSS (Table A4), plotted against (1) PO_4-P concentration and (2) total soluble inorganic nitrogen concentration for each treatment. These graphs are helpful in determining nutrient availability in a sample, as well as supporting the determined limiting nutrient. The maximum growth is compared to a theoretical growth vs. concentration curve derived from the AAM control. According to theory, for a certain concentration of nutrient (N or P), there should be a predictable increase in biomass over time. If a particular nutrient is deficient then growth will be below the expected level as predicted by the AAM control. Growth may also be too low if the nutrients are in an unavailable form even through chemical analysis indicated adequate concentrations of all nutrients in the test water.

Because of the difficulty of measuring biomass in low density cultures, relative fluorescence of in vivo chlorophyll a was used to estimate biomass during the early phases of growth. Calculations of

average maximum specific growth rate batch ($\hat{\mu}_b$) were made using these numbers. All $\hat{\mu}_b$ values were observed to occur between days 1 and 4 of the growth curve. Maximum relative fluorescences ($\hat{X}$) were also determined. Both the $\hat{\mu}_b$ and $\hat{X}$ values are tabulated in Table A5. Even though relative fluorescence measures a physiological response and optical density measures a standing crop (biomass) response, the responses should correlate relative to the biomass; this is shown by the following regression equation which relates maximum observed RF and maximum observed OD ($r^2 = 0.946$):

$$OD = 0.0017 \ (RF \times 30) + 0.045 \qquad (A2)$$

Results

1. Wild Rice River at Twin Valley Gauge (Figures A1-A2)

Based upon chemical analysis alone it was difficult to determine the status of this water. Indigenous nitrogen and phosphorus concentrations were moderately high and in a proportion (N/P ratio = 26.8) that would make this water somewhat fertile without treatment. A slight degree of phosphorus limitation may be possible but this is hard to predict. Generally waters with N/P ratios (all N/P ratios in the text refer to TSIN/OP) $< \sim 15$ are nitrogen limited while those greater are phosphorus limited. Values near 15 are optimal if the phosphorus concentration is ≥ 0.010 mg P/ℓ. However, Porcella et al. (1970) showed N/P ratios varying in the range of 6 to 88 to be optimal or nonoptimal depending on the test water involved; therefore, it can only be assumed that phosphorus is limiting in this case.

As was predicted, bioassay indicated a significant increase in biomass in the untreated sample (Table A4 - 64.8 mg/ℓ VSS in Treatment A). The maximum specific growth rate was high for all treatments of Sample 1 indicating that the algae are quickly using the available nutrients from day 0 to day 4, then growth rate rapidly decreases when nutrients are exhausted. Increased biomass in Treatments A, B, E, and F showed that indigenous nutrients were utilized, but the addition of phosphorus (Treatment C) created a lower total N/P ratio (N/P = 10.0),

and the system was able to use all the remaining indigenous nitrogen. This is a clear indication of a phosphorus limitation. Figures A1 and A2 point out that the phosphorus limitation was partially due to the small increase in growth observed over Treatments A, B, and F and higher growth in Treatment D.

2. Inflow to Lower Rice Lake (Figures A3-A4)

The conditions in this test water indicated nitrogen limitation. The chemical analysis showed low nutrient levels in the untreated water. The initial N/P ratio was 8.6, a level which predicts nitrogen limitation, although slight. The bioassay confirms slight nitrogen limitation as visualized in Figures A3 and A4. Biomass in Treatment B increased only slightly because phosphorus rapidly becomes limiting due to the initially low levels of all nutrients. There is a delicate line between which nutrient may be limiting at any one time. At this particular sampling time it appears nitrogen was limiting.

It will be noted that the specific growth rate (Table A5) in Treatment B increased significantly over the untreated sample. Treatments D and G show even greater increases in $\hat{\mu}_b$, further confirming that nitrogen is only partially limiting in this sample.

3. Wild Rice River, Upper End Conservation Pool (Figures A5-A6)

The observed results in the upper end conservation pool sample were slightly different from results in the previous samples. The chemical analyses indicated possible phosphorus limitation (N/P = 29.1), but as with the previous samples the limitation appeared to be only partial. Figures A5 and A6 show that bioassay confirmed this limitation. The maximum biomass observed was 32.9 mg/ℓ as VSS in the untreated sample while the maximum biomass increased to 111.0 mg/ℓ when phosphorus was added (Treatment C). When N is added in addition to P (Treatment D) greater response occurred indicating N is also limiting.

The unique point to note in this sample was the observed response when both trace elements and HCO_3^- were added to the system. This indicated that trace elements and carbon were also limiting. Since *Selenastrum capricornutum* derives carbon from CO_2 in the atmosphere, it is doubtful that carbon limitation is involved. Since HCO_3^- spikes

add buffer capacity to the sample as well as an additional carbon source, this aspect may also be considered. This response may result from interaction of trace elements and phosphorus with calcium in samples having high hardness. Added bicarbonate helps prevent precipitation of such complexes making phosphorus more available.

4. Dayton Hollow Dam Mid Pool, Depth Integrated (Figures A7-A8)

It was concluded from bioassay and to a lesser degree chemical analysis that both nitrogen and phosphorus were limiting. The untreated sample was somewhat fertile as indicated by (1) the optimal N/P ratio of 21.9, (2) an observable increase in biomass in Treatment A (Table A4), (3) the high maximum specific growth rate (Table A5), and (4) Figures A7 and A8 which graphically represent the growth. Only when both nitrogen and phosphorus (Treatment D) and AAM (Treatment G) were added did the biomass as well as the $\hat{\mu}_b$ values increase over the untreated sample.

5. Ottertail River, Inlet to Dayton Hollow (Figures A9 and A10)

Results for the Ottertail River are identical to results observed in Sample 4 above: both N and P are limiting. Samples 4 and 5 are identical in respect to all aspects observed during the bioassay. The indigenous levels of N and P were the same, the growth response to treatments the same, and the maximum growth rates in all treatments were remarkably similar.

Nutrient Availability

In order to ascertain N and P availability, maximum growth was plotted against N and P concentrations present in the various treatments (Figures A11-A20) for each of the samples.

1. Wild Rice River at Twin Valley Gauge (Figures A11 and A12)

This water showed equal yields (mg VSS/mg TSIN and mg VSS/mg PO_4) for N and P in the untreated sample when compared to algae grown in AAM (AAM represents the optimum conditions for growing *S. capricornutum*). This indicated that all the N and P was available in the untreated sample. Addition of phosphorus (Treatment C, Figure A11)

increased growth until nitrogen became limiting. Nitrogen, when added (Treatment B, Figure A12), created no increase in biomass. When both N and P were added, increase in biomass equals the AAM control. This indicated that both TSIN and OP were utilized to yield biomass without inhibition.

2. Inflow to Lower Rice Lake (Figures A13 and A14)

Both TSIN and OP were available in this sample to produce biomass. Algae utilized completely the N and P in Treatments A, D, and G. When P was added alone, no increase in biomass occurred and only a slight increase occurred when N was added alone. This indicated a partial N limitation, but an eventual limitation by both N and P.

3. Wild Rice River Upper End Conservation Pool (Figures A15 and A16)

Growth in the untreated sample fell slightly below the expected level indicating a scant unavailability in the untreated sample. Addition of P (Figure A15, Treatment C) increased biomass until N became limiting. Both N and P when added brought about maximum growth. It appeared that nutrients were in available form in this sample.

4. Dayton Hollow Dam, Mid Pool, Depth Integrated (Figures A17 and A18)

5. Ottertail River, Inlet to Dayton Hollow (Figures A19 and A20)

N and P limitation is confirmed by no increased biomass when either was added alone. When both were added growth was equal to AAM indicating total nutrient availability.

Conclusions

1. Samples 1 and 3 were limited partially by phosphorus, but nitrogen became limiting as well. This fact was supported by a slight increase in growth when phosphorus was added but a much greater increase when both N and P were added.

2. Sample 2 exhibited nitrogen limitation, but the same situation was true here as with Samples 1 and 3. Nitrogen was only partially limiting while phosphorus also became limiting when the P reserve was lowered.

3. Samples 4 and 5 were definitely limited by both nitrogen and phosphorus.

4. There was a slight degree of trace element limitation in Sample 3. Carbon limitation was possible but not likely.

5. Ranking of bioassay response in untreated samples showed the following (greatest to least).

1 > 5 > 4 > 3 > 2

When N and P were increased by spiking the ranking was:

3 > 1 > 4 > 5 > 2

6. Relative fluorescence response correlated well with optical density response. This allows growth rate data to be analyzed along with biomass. Using both parameters, N/P ratios were essentially confirmed.

7. No toxicity was observed.

RUN 2 (JULY 1978)

Five samples arrived at the Utah Water Research Laboratory by air freight from Fargo, North Dakota, on July 13, 1978. The samples were in polyethylene containers and on partial ice as in April 1978. The samples were subjected to pretreatment identical with bioassay, run 1. Samples 4 and 5, although from the same sites, were labeled somewhat differently. The sites included:

1. Wild Rice River at Twin Valley Gauge
   11 July 1978
2. Inflow to Lower Rice Lake
   11 July 1978
3. Wild Rice River Upper End Conservation Pool
   11 July 1978
4. Dayton Hollow Reservoir, Ottertail River
   11 July 1978
5. Ottertail River Upstream of Dayton Hollow
   11 July 1978

Sample spikes or treatments are the same as those in bioassay run 1 (page A3). All conditions, sample volume, temperature, and lighting, mimic the April 1978 bioassay.

Results

In general, the test waters of July 1978 have taken on characteristics somewhat different from before. Initial chemical analysis showed much lower concentrations of indigeneous nitrogen. Phosphorus concentrations also varied but in a much more random fashion. Samples 1 and 3 had lower P values while 2 and 5 were higher. The phosphorus concentration of Sample 4 increased drastically.

Due to the changes in nutrient concentrations, all test waters were ascertained to be nitrogen limited based on chemical analysis. N/P ratios are listed in Table A7. The overall lower nutrient concentrations at this time suggested less fertile samples than noted in April 1978.

1. Wild Rice River at Twin Valley Gauge (Figures A21 and A22)

The concentrations of indigenous nutrients appeared to be too low to support any kind of growth in the untreated sample. Generally a test water must contain $\geq$ 0.010 mg P/ℓ if growth is to result. The level of 0.016 mg P/ℓ in the Wild Rice River appeared to be less than adequate. Because of this fact, the addition of N (Treatment B) had no effect. This sample was both nitrogen and phosphorus limited. Maximum growth rates (Table A8) are substantially higher when both N and P were made available.

2. Inflow to Lower Rice Lake (Figures A23 and A24)

The N/P ratio of 1.6 clearly indicated nitrogen limitation and treatment with nitrogen (Treatment B) confirmed this assumption. Figures A23 and A24 visually represent the increased response upon addition of nitrogen. The $\hat{\mu}_b$ values are extremely low except in treatments which included nitrogen (Treatments B, D, and G), also confirming nitrogen limitation.

Phosphorus will become limiting as indicated by the subnormal, although increased growth in Treatment B. The situation created in Treatment B is one of phosphorus limitation (N/P = 44.4). The relatively low level of indigenous phosphorus was exhausted before maximum growth could be reached.

3. Wild Rice River, Upper End Conservation Pool (Figures A25 and A26)

The sample responded to nitrogen addition with increased biomass but not significantly enough to classify the sample as N limited. The sample was identified as both N and P limited. Indigenous nutrients were at a level that was considered relatively low, making the sample infertile in an untreated form. The sample responds well to Treatments D and G with greatly increased maximum specific growth rates.

4. Dayton Hollow Reservoir, Ottertail River (Figures A27 and A28)

(Dayton Hollow Dam, Mid Pool, Depth Integrated)

A different situation arose upon examination of this test water. It was assumed from chemical analysis that nitrogen was the limiting factor due to the high concentration of phosphorus (100 $\mu g/\ell$). It appears that nitrogen does play a role in growth limitation, but other nutrients must also be considered. Figures A27 and A28 indicate that the addition of nitrogen enhanced growth. This fact was substantiated by the maximum biomass observed (Table A9). However, this increase was far below what was predicted. The initial N/P ratio of 3.2 was increased to 24.2 upon addition of N in Treatment B. An N/P ratio of 24.2 is within the optimum growth range; therefore, it was assumed growth would be near optimum. Addition of both N and P in Treatment D lowered the N/P ratio to 12.5 (still within the optimum range), but growth remained at a subnormal level. It was assumed a trace element or HCO_3^- limitation in conjunction with the N limitation was involved because growth in AAM (Treatment G) was normal.

5. Ottertail River Upstream of Dayton Hollow (Figures A29 and A30)

(Ottertail River Inlet to Dayton Hollow)

Since the April bioassay the phosphorus concentration doubled and the nitrogen concentration was only one-third its previous level. This created a definite nitrogen limitation in the test water. Bioassay was in agreement with this assumption made on the chemical analysis alone. Nitrogen addition caused the $\hat{\mu}_b$ value to increase from 1.07 to 1.46 and the biomass to increase from 36.4 to 72.1. Phosphorus limitation also played a part as indicated by further biomass increase (122.5) when P was added along with N (Treatment D).

Nutrient Availability

1. Wild Rice River at Twin Valley Gauge (Figures A31 and A32)
2. Inflow to Lower Rice Lake (Figures A33 and A34)
3. Wild Rice River Upper End Conservation Pool (Figures A35 and A36)

Both nitrogen and phosphorus limitation was confirmed. N or P additions have no effect on the total biomass; it remained the same as the untreated sample. As indicated by the growth equaling AAM, nutrients were in an available form for both samples.

4. Dayton Hollow Reservoir, Ottertail River (Figures A37 and A38)
5. Ottertail River Upstream of Dayton Hollow (Figures A39 and A40)

Nitrogen limitation is pointed out well using the mg VSS/mg PO_4 plots. Figures A33, A37, and A39 show that growth in the untreated samples was subnormal for the level of indigenous phosphorus, but according to Figures A34, A38, and A40, biomass was greater than normal for the level of indigenous nitrogen in the samples. This clearly indicates complete usage of nitrogen with an excess of phosphorus. When both N and P were added to Samples 2 and 5, biomass was near normal, indicating that all nutrients were available.

The situation in Sample 4 is somewhat different. Due to the trace metal and HCO_3^- limitation, growth increase was minimal when N and P were added. Macronutrients (N and P) were not made available until trace metals and HCO_3^- were added in Treatment G.

Conclusions

1. Samples 1 and 3 contained extremely small concentrations of both nitrogen and phosphorus; therefore, the untreated samples tended to be infertile. Only upon addition of both nitrogen and phosphorus did biomass increase.

2. Samples 2 and 5 also exhibited infertile tendencies, but the addition of nitrogen lessened the infertility to some degree. However, the indigenous P as well as N was relatively low, making these samples subject to phosphorus limitation as well.

3. Sample 4 had several limiting factors. Nitrogen was definitely one of those factors, but trace element or HCO_3^- limitation was also involved.

4. Ranking of the untreated sample response showed the following:

$5 > 4 > 2 > 3 > 1$

When N and P were increased by spiking the ranking was:

$5 > 2 > 3 > 1 > 4$

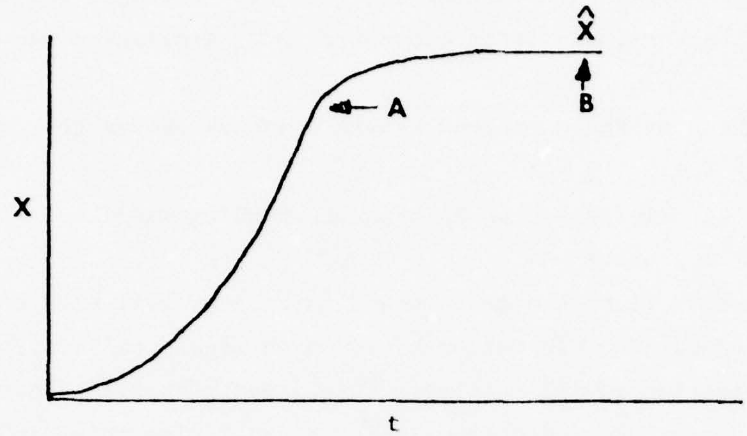
5. Relative fluorescence response correlates well with optical density response. The N/P ratios based on chemical analysis indicated nitrogen limitation in all samples. This appears correct; however, due to the low nutrient concentrations, both N and P play roles in limitation.

6. No toxicity was observed.

A BRIEF REVIEW OF ALGAL GROWTH DYNAMICS

The growth of algae is limited by the quantity of environmental factors such as light, temperature, and nutrients. When light, temperature, and other environmental conditions are at standardized levels as in the algal assay method (EPA 1971), the bioassay shows responses related to nutrient concentrations. Previous studies have shown (e.g., Porcella et al. 1970) that two parameters of growth can be related to the concentration of limiting nutrient, specific growth rate, and maximum growth. Actual measurements of growth from which growth parameters are calculated usually require a variety of methods (for example, cell counts, cell volume, chlorophyll, turbidity, optical density, dry weight of cells, particulate carbon), limited only by economics and technician time. The limiting nutrient is that nutrient (required element) which is in lowest concentration compared to all other nutrients in relation to the needs of the cell.

The growth parameters can be calculated or estimated from measurements of the growth (X, cells) of an algal population over a period of time (t, days):



The maximum growth ($\hat{X}$) observed occurs when the limiting nutrient is exhausted and cells not only cease dividing but cease increasing in mass. Operationally, the maximum observed value of the growth measurement is used and the day of the observation is recorded as follows: $\hat{X}_B$ (t = B). Populations increase geometrically (e.g., doubling at specific time increments) and the rate of change can be defined as:

$$\frac{dX}{dt} = \mu X; \qquad \mu = \frac{dX}{X\,dt} \tag{A3}$$

Where μ (days^{-1}) is the specific growth rate. This applies only over a limited range (i.e., where the limiting nutrient has not been completely utilized). Integrating over time (from t = 0 to t = t) for a growing population starting from known cell concentration (X_i) to a greater cell concentration (X_{i+t}), one obtains: $X_{i+t} = X_i\, e^{\mu t}$ Operationally, μ is determined either by plotting X on semilog paper and determining the slope from the straight line portion of the curve (up to about point A) or by using the maximum observed slope in the entire growth curve ($\hat{\mu}_b$).

The growth parameters are related to the limiting nutrient concentration (S) by specific growth functions (for review see Porcella et al. 1970; EPA 1971). Maximum growth ($\hat{X}$) is related to the limiting nutrient by the concept of yield (Y) (i.e., mass of cell growth per unit of nutrient mass):

$$\hat{X} = YS_o; \quad Y = \frac{\hat{X}}{S_o} \tag{A4}$$

where S_o is the initial concentration of limiting nutrient and it is assumed (on good evidence) that essentially all of the nutrient is utilized during cell growth.

Growth rate is limited by catalysis, and enzyme kinetics have been assumed limiting as follows:

$$\mu = \frac{\hat{\mu}S}{K_s + S} \tag{A5}$$

where $\hat{\mu}$ is the maximum physiological population growth rate and K_s is the half-saturation coefficient. Data in this report were analyzed only in terms of yield because measurement techniques used were not sensitive enough to determine algal growth at the low cell concentrations where growth increases the fastest.

When samples are bioassayed, growth is measured by an appropriate method and the growth parameters determined. These parameters in themselves may be insufficient to determine: (a) the limiting nutrient, (b) the effects of increases in specific nutrients, and (c) the interactions between nutrient additions and chemical composition and interactions of the sample. Nutrient additions are made (termed "spiking") and bioassay responses determined; an example is the "Protocol for Nutrient Spiking" in Table A2. Spiking effects on growth parameters can be used to determine the limiting nutrient for that sample and the effects of interactions of the increase of the specific compound or element. For example, when a specific nutrient addition causes a large response in a sample and other additions do not, that specific nutrient is defined as limiting algal growth for the test conditions and bioassay organism.

High hardness and alkalinity are found in waters draining limestone geological basins. The removal of CO_2 from the inorganic carbon pool of alkalinity causes an increase in pH and a shift to carbonate ions which then precipitate calcium ions from the hardness complement. Calcium phosphates and metal hydroxides also are removed along with the

$CaCO_3$ precipitates and become unavailable for algal growth. The addition of bicarbonate to such hard water systems increases the alkalinity pool and leads to better pH control making phosphates and trace metals more available. In any case, high hardness-alkalinity waters which appear to be limited by nitrogen or other components based on chemical analysis can, during the bioassay, and presumably in nature, become limited by phosphorus or trace metals because of precipitation.

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Table A1. Chemical analyses and results.[a]

Parameter	Units	Method	Refs.*	1	2	3	4	5
PO_4-P	μg/ℓ	Antimony Molybdate; Ascorbic Acid	1	55	25	51	37	41
Total Soluble P (TSP)	μg/ℓ	Persulfate Digestion	1	87	55	79	61	65
$(NO_3 + NO_2)$-N	μg/ℓ	Automated Cadmium; Reduction	2	1280	160	1290	630	590
NH_3-N	μg/ℓ	Indophenol	3	194	56	194	180	154
Total Soluble Inorganic Nitrogen (TSIN)	μg/ℓ			1474	216	1484	810	744

* 1. Standard Methods 1971.
2. Environmental Protection Agency. 1974.
3. Solorzano. 1969.

[a]UWRL is an EPA-certified lab through the Utah State Division of Health, Salt Lake City.

Table A2. Algal assay medium (AAM).*

	Compound	Concentration in NAAM: Compound mg/ℓ	Concentration in NAAM: Element mg/ℓ	
A_1	$NaNO_3$	25.500	N	4.2
A_2	$MgCl_2 \cdot 6H_2O$	12.171	Mg	2.9
	$MgSO_4 \cdot 7H_2O$	14.700		
A_3	$CaCl_2 \cdot 2H_2O$	4.410	Ca	1.2
A_4	$NaHCO_3$	15.000		
B	K_2HPO_4	1.044	P	0.186
		μg/ℓ	μg/ℓ	
C	H_3BO_3	185.64	B	32.45
	$MnCl_2 \cdot 4H_2O$	417.18	Mn	115.80
	$ZnCl_2$	32.70	Zn	15.68
	$Na_3MoO_4 \cdot 2H_2O$	7.26	Mo	2.88
	$CoCl_2 \cdot 6H_2O$	1.43	Co	0.35
	$CuCl_2 \cdot 2H_2O$	0.01	Cu	0.004
D	$FeCl_3 \cdot 6H_2O$	160	Fe	33.05
	$Na_2EDTA \cdot 2H_2O$	300		
			mg/ℓ	
Protocol for Nutrient Spiking			S	1.91
A_1	Nitrogen		Na	11.04
B	Phosphorus		K	0.47
A_1 + B	N + P		C	2.14
C + D	Trace Elements (TE)			
ALL	NAAM			

* (EPA 1971).

Table A3. Maximum amount of growth observed as optical density, 750 nm, 1 cm.

Sample	Treatment A (Sample only)	Treatment B (NO_3-N)	Treatment C (PO_4-P)	Treatment D (NO_3-N + PO_4-P)	Treatment E (Trace Elements)	Treatment F (HCO_3^-)	Treatment G[a]	N/P Ratios[b] TSIN/OP	N/P Ratios[b] TSIN/TSP
1	0.174	0.150	0.248	0.404	0.169	0.175	0.393	27	17
2	0.070	0.076	0.050	0.340	0.060	0.064	0.337	9	4
3	0.084	0.120	0.307	0.433	0.166	0.168	0.343	29	19
4	0.100	0.096	0.137	0.389	0.110	0.109	0.350	22	13
5	0.119	0.095	0.130	0.371	0.094	0.102	0.359	18	11
AAM							0.276	23	23

[a]Treatment G = 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of trace elements, HCO_3^-, $CaCl_2$, and $MgSO_4$.

[b]N/P Ratio = Nitrogen/Phosphorus Ratio

TSIN = Total Soluble Inorganic Nitrogen

OP = Orthophosphate

TSP = Total Soluble Phosphorus.

Table A4. Maximum amount of growth observed, mg/ℓ VSS.[a]

Sample	Treatment A (Sample only)	Treatment B (NO_3-N)	Treatment C (PO_4-P)	Treatment D (NO_3-N + PO_4-P)	Treatment E (Trace Elements)	Treatment F (HCO_3^-)	Treatment G[b]
1	64.8	56.0	90.3	146.7	62.7	64.8	141.1
2	28.0	30.1	21.0	122.5	24.5	25.9	121.5
3	32.9	45.5	111.0	155.1	61.6	62.3	123.6
4	38.5	37.1	51.5	139.7	42.0	41.7	126.0
5	45.2	36.8	49.0	133.4	36.4	39.2	129.2
AAM							90.1

[a]VSS = Volatile Suspended Solids
VSS, mg/ℓ = 350 (Optical Density) + 3.5 (Porcella et al. 1973).

[b]Treatment G = 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of trace elements, HCO_3^-, $CaCl_2$, and $MgSO_4$.

Table A5. Parameters estimated from relative fluorescence (RF × 30).

Sample	Parameter*	Treatment A	B	C	D	E	F	G
1	$\hat{\mu}_b$, days^{-1}	1.12	1.26	1.31	1.18	1.10	1.24	1.51
	$\hat{X}$ (day), RF × 30	890(5)	890(5)	1120(5)	2030(5)	870(5)	830(5)	2160(7)
2	$\hat{\mu}_b$	0.60	0.95	0.34	1.26	0.57	0.59	1.25
	$\hat{X}$	146(3)	347(5)	87(3)	1290(5)	151(5)	145(4)	1270(7)
3	$\hat{\mu}_b$	0.81	1.05	1.39	1.53	1.16	1.24	1.46
	$\hat{X}$	423(5)	583(5)	1160(5)	1930(7)	690(5)	770(5)	1810(5)
4	$\hat{\mu}_b$	1.12	1.07	1.24	1.43	1.12	1.10	1.46
	$\hat{X}$	397(5)	427(5)	370(3)	1490(5)	423(5)	393(5)	1500(5)
5	$\hat{\mu}_b$	1.16	1.12	1.24	1.44	1.17	1.09	1.48
	$\hat{X}$	460(5)	463(5)	360(3)	1510(7)	447(5)	407(7)	1530(5)
AAM Control	$\hat{\mu}_b$							1.30
	$\hat{X}$							1330(7)

* $\hat{\mu}_b$, days^{-1} = maximum specific growth rate.

$\hat{X}$ (day), RF × 30 = maximum relative fluorescence reading (day of occurrence).

Table A6. Chemical analyses and results.[a]

Parameter	Units	Method	Refs.*	1	2	3	4	5
Orthophosphate (PO_4-P)	μg/ℓ	Antimony Molybdate; Ascorbic Acid	1	16	49	33	100	82
Total Soluble P (TSP)	μg/ℓ	Persulfate Digestion	1	28	98	59	192	157
Nitrate + Nitrite [(NO_3 + NO_2)-N]	μg/ℓ	Automated Cadmium; Reduction	2	50	50	130	170	150
Ammonia (NH_3-N)	μg/ℓ	Indophenol	3	37	28	25	150	87
Total Soluble Inorganic Nitrogen (TSIN) [(NO_3+NO_2+NH_3)-N]	μg/ℓ			87	78	155	320	237

* 1. Standard Methods 1971.

2. Environmental Protection Agency. 1974.

3. Solorzano. 1969.

[a]UWRL is an EPA-certified lab through the Utah State Division of Health, Salt Lake City.

Table A7. Maximum amount of growth observed as optical density, 750 nm, 1 cm.

Sample	Treatment A (Sample only)	Treatment B (NO_3-N)	Treatment C (PO_4-P)	Treatment D (NO_3-N + PO_4-P)	Treatment E (Trace Elements)	Treatment F (HCO_3^-)	Treatment G[a]	N/P Ratios[b]	
								TSIN/OP	TSIN/TSP
1	0.029	0.037	0.033	0.304	0.030	0.024	0.300	5	3
2	0.043	0.126	0.038	0.323	0.044	0.039	0.326	2	< 1
3	0.039	0.057	0.040	0.317	0.047	0.042	0.307	5	3
4	0.047	0.126	0.039	0.102	0.060	0.056	0.351	3	2
5	0.094	0.196	0.058	0.340	0.060	0.074	0.318	3	2
AAM							0.251	23	23

[a]Treatment G = 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of trace elements, HCO_3^-, $CaCl_2$, and $MgSO_4$.

[b]N/P Ratio = Nitrogen/Phosphorus Ratio

TSIN = Total Soluble Inorganic Nitrogen

OP = Orthophosphate

TSP = Total Soluble Phosphorus.

Table A8. Parameters estimated from relative fluorescence (RF × 30).

Sample	Parameter *	Treatment						
		A	B	C	D	E	F	G
1	$\hat{\mu}_b$, days^{-1}	0.34	0.41	0.39	1.36	0.37	0.36	1.42
	$\hat{X}$ (day), RF x 30	92(5)	155(5)	90(5)	1560(5)	78(5)	67(5)	1390(5)
2	$\hat{\mu}_b$	0.19	0.92	0.11	1.14	0.19	0.17	1.18
	$\hat{X}$	139(5)	647(5)	114(5)	1800(6)	106(5)	125(5)	1920(5)
3	$\hat{\mu}_b$	0.41	0.54	0.45	1.52	0.65	0.61	1.36
	$\hat{X}$	142(5)	204(5)	145(5)	1420(6)	147(5)	164(5)	1490(7)
4	$\hat{\mu}_b$	1.00	1.02	0.54	1.23	0.81	0.88	1.60
	$\hat{X}$	334(5)	860(6)	227(5)	1800(6)	137(3)	379(5)	2027(5)
5	$\hat{\mu}_b$	1.07	1.46	0.70	1.52	0.80	0.90	1.66
	$\hat{X}$	265(4)	1090(7)	127(6)	1750(5)	121(5)	174(5)	1930(7)
AAM Control	$\hat{\mu}_b$							1.70
	$\hat{X}$							1710(7)

* $\hat{\mu}_b$, days^{-1} = maximum specific growth rate.

$\hat{X}$ (day), RF × 30 = maximum relative fluorescence reading (day of occurrence).

Table A9. Maximum amount of growth observed, mg/ℓ VSS.[a]

Sample	Treatment A (Sample only)	Treatment B (NO_3-N)	Treatment C (PO_4-P)	Treatment D (NO_3-N + PO_4-P)	Treatment E (Trace Elements)	Treatment F (HCO_3^-)	Treatment G[b]
1	13.7	16.5	15.1	109.9	14.0	11.9	108.5
2	18.6	47.6	16.1	116.6	18.9	17.2	117.6
3	17.2	23.5	17.2	114.5	16.1	18.2	111.0
4	20.0	50.4	17.2	39.2	24.5	23.1	126.4
5	36.4	72.1	23.8	122.5	24.5	29.4	114.8
AAM							91.4

[a] VSS = Volatile Suspended Solids
VSS, mg/ℓ = 350 (Optical Density) + 3.5 (Porcella et al. 1973).

[b] Treatment G = 2.1 mg/ℓ NO_3-N + 0.093 mg/ℓ PO_4-P + AAM levels of trace elements, HCO_3^-, $CaCl_2$, and $MgSO_4$.

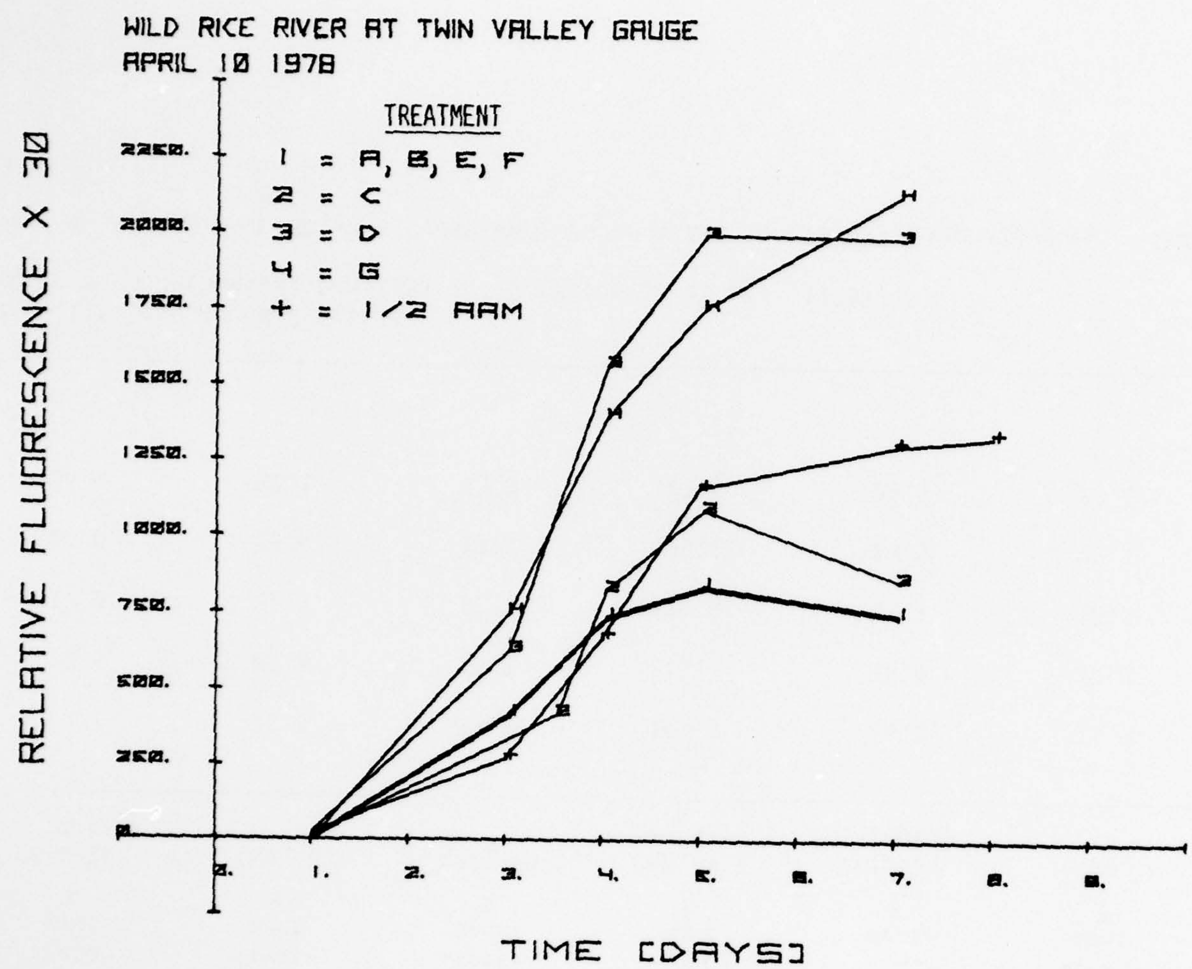


Figure A1. Relative fluorescence of sample 1 (April 10) with various treatments

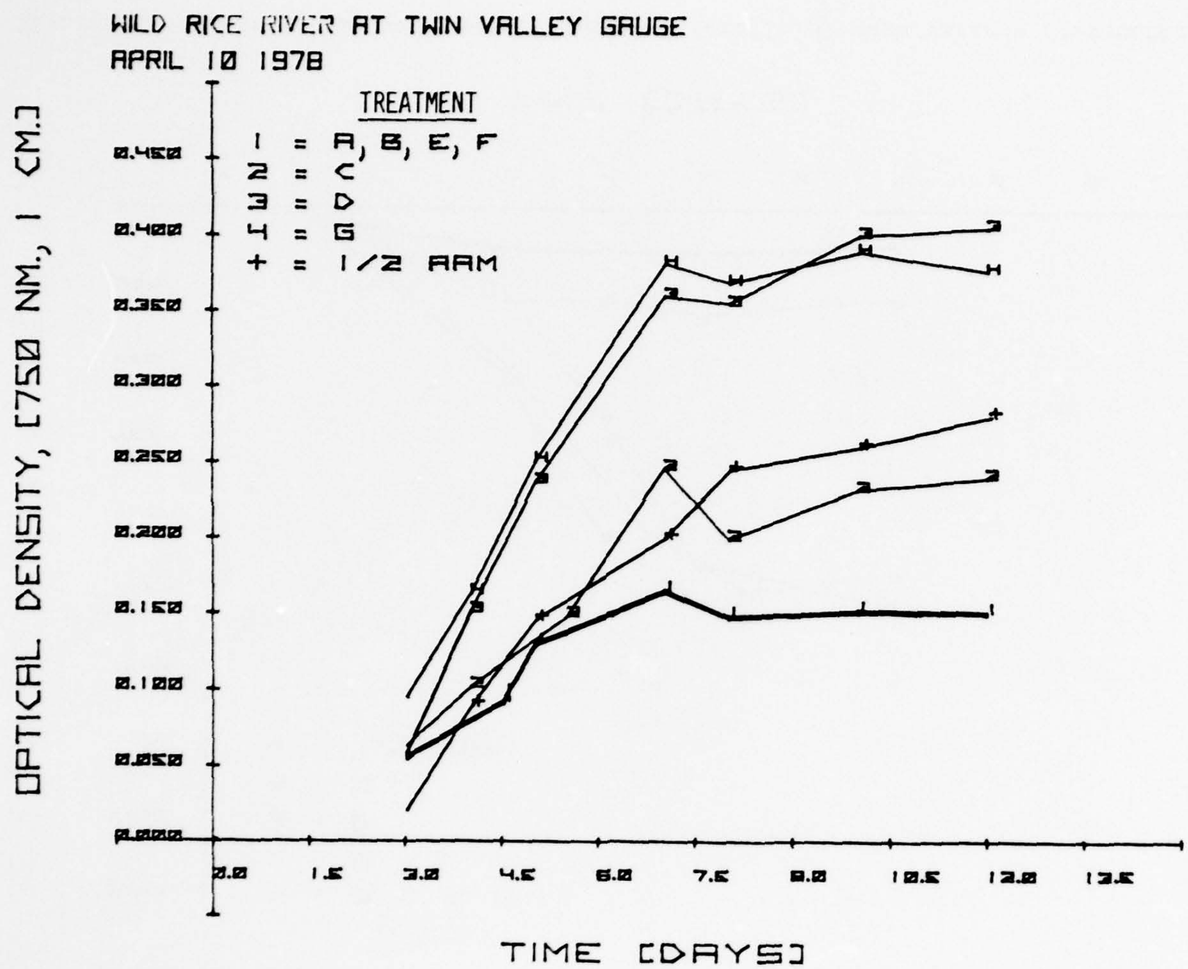


Figure A2. Optical density of sample 1 (April 10) with various treatments

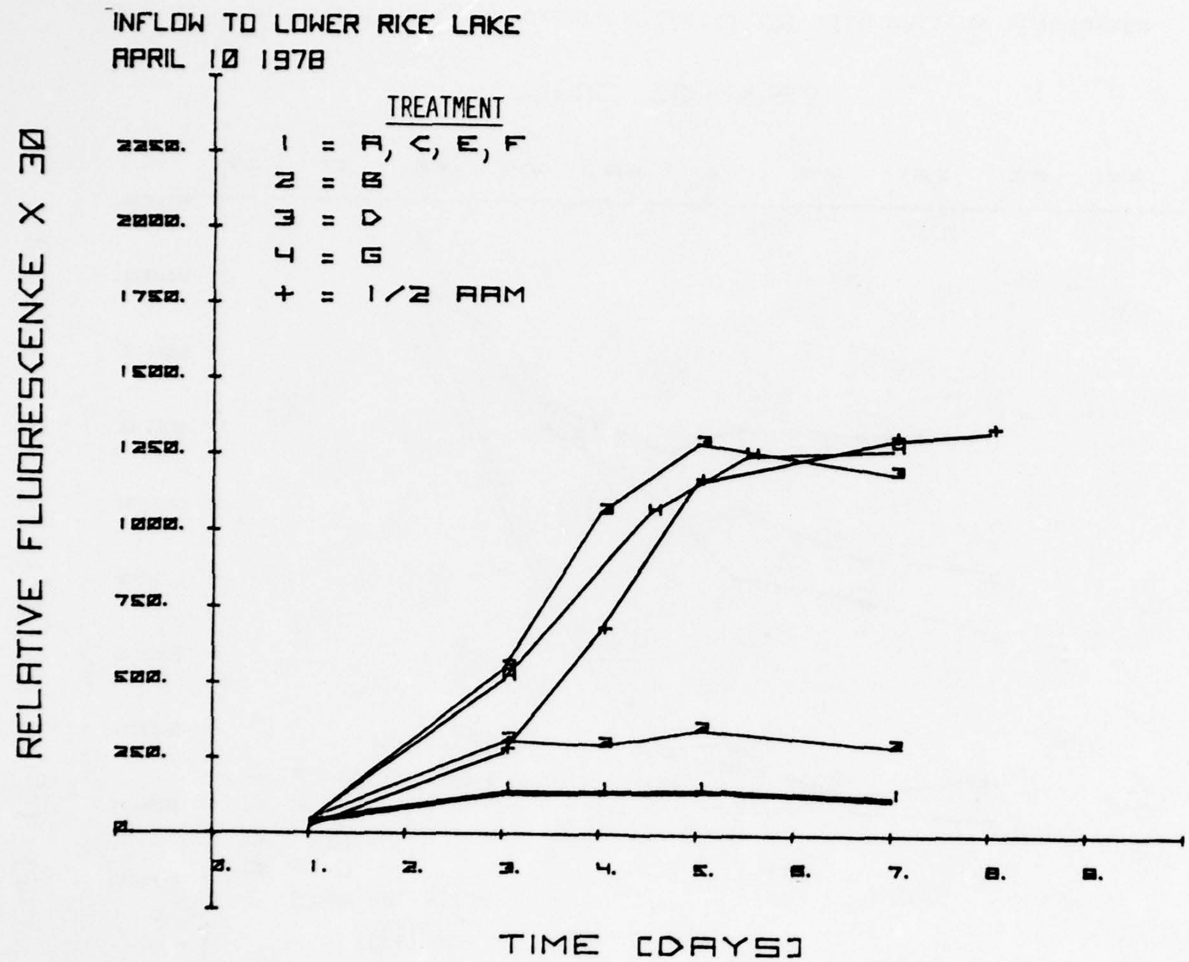


Figure A3. Relative fluorescence of sample 2 (April 10) with various treatments

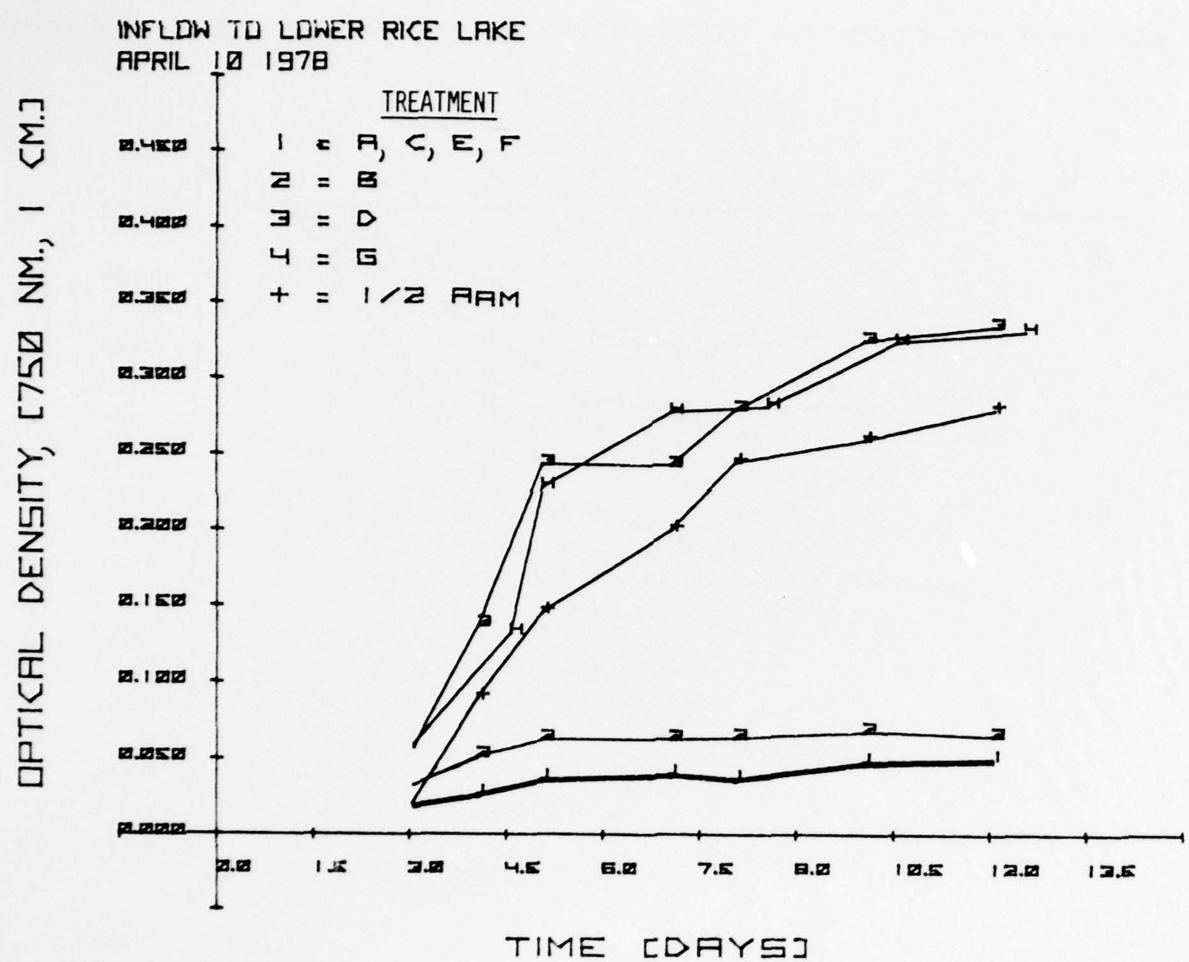


Figure A4. Optical density of sample 2 (April 10) with various treatments

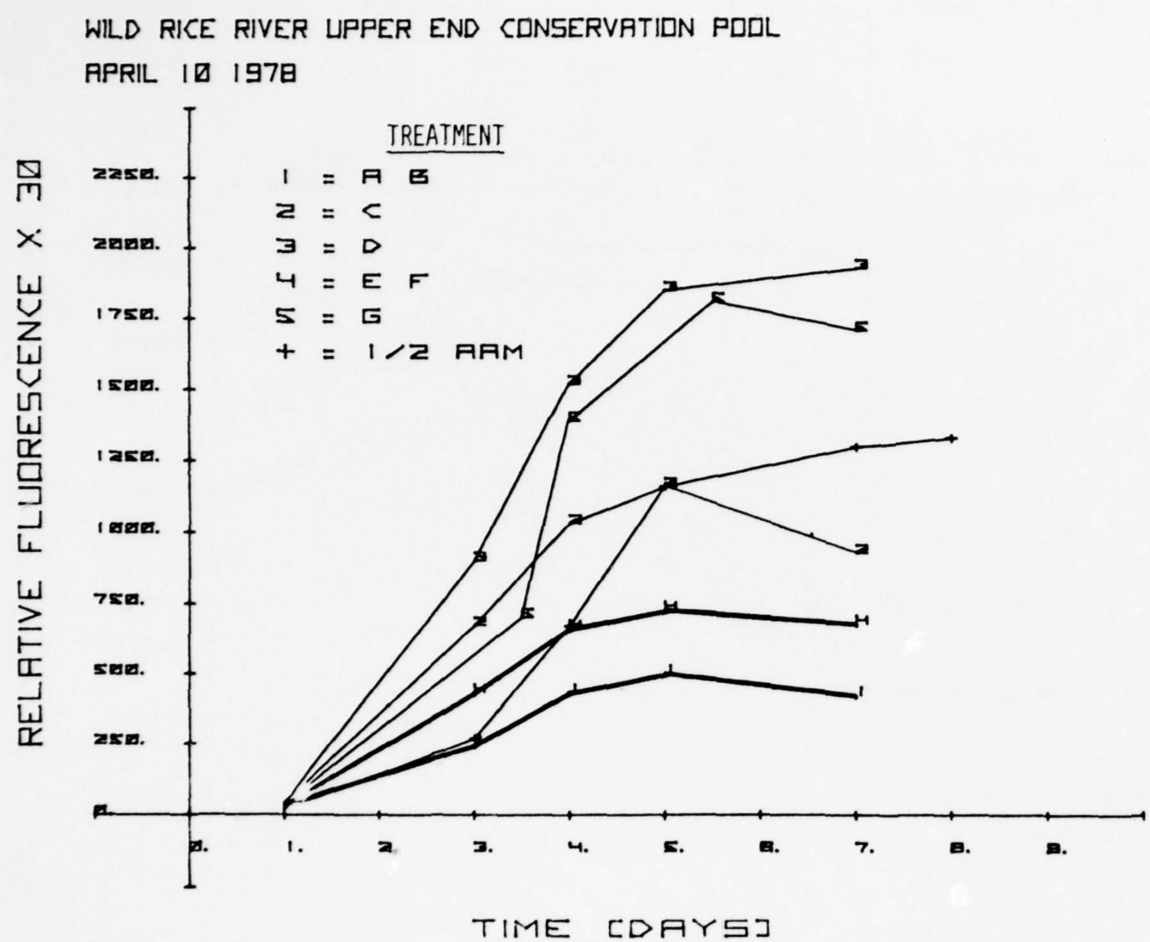
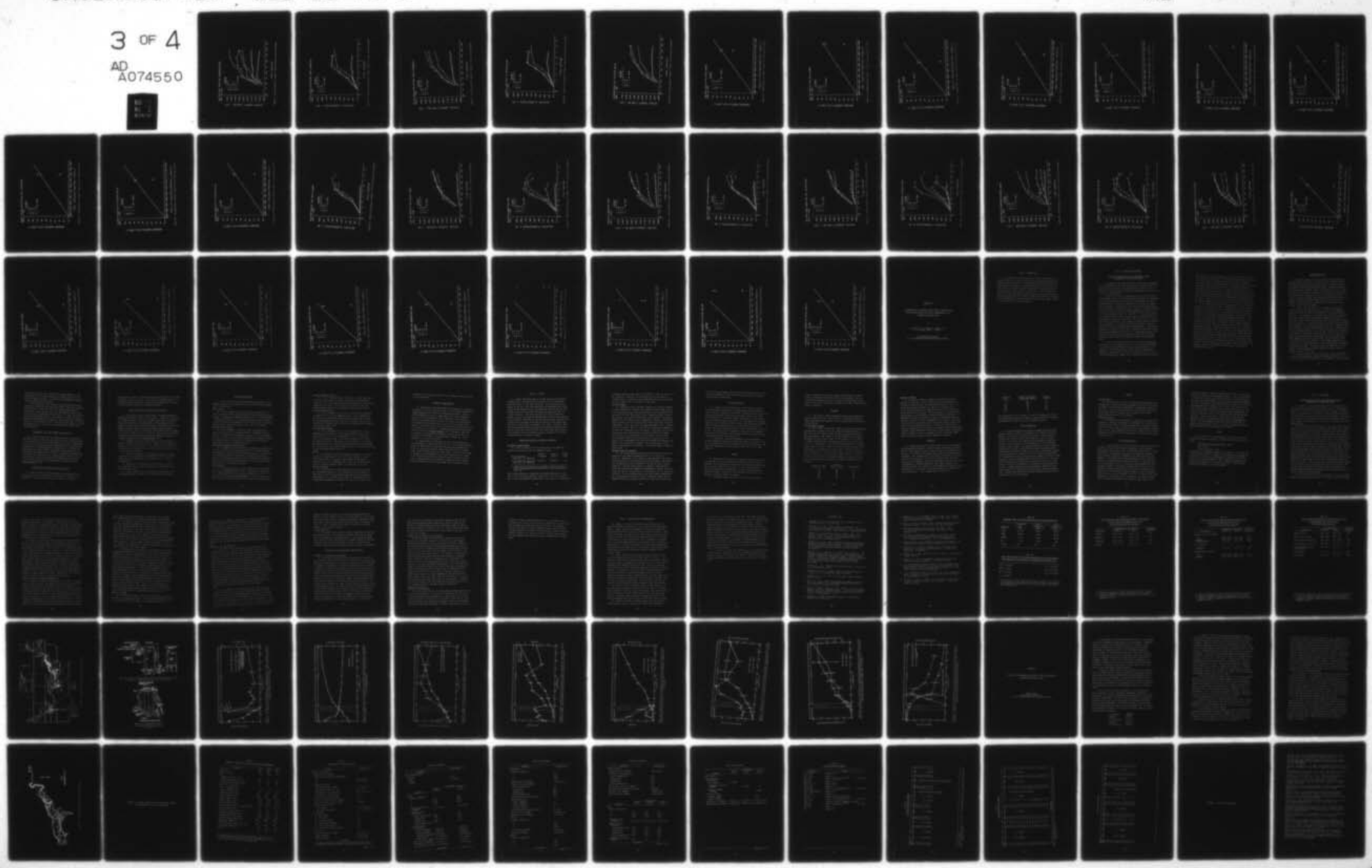


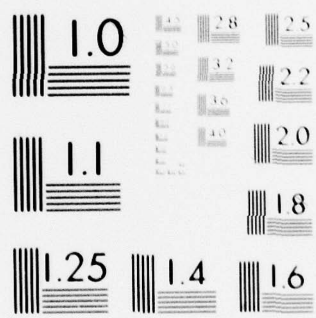
Figure A5. Relative fluorescence of sample 3 (April 10) with various treatments

AD-A074 550 ARMY ENGINEER WATERWAYS EXPERIMENT STATION VICKSBURG--ETC F/G 13/2
WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE, WILD RIC--ETC(U)
JUL 79 D E FORD, K W THORNTON, W B FORD

UNCLASSIFIED WES-EL-79-5 NL

AD
A074550




1.0
2.8
2.5
3.2
2.2
3.6
4.0
2.0
1.1
1.8
1.25
1.4
1.6

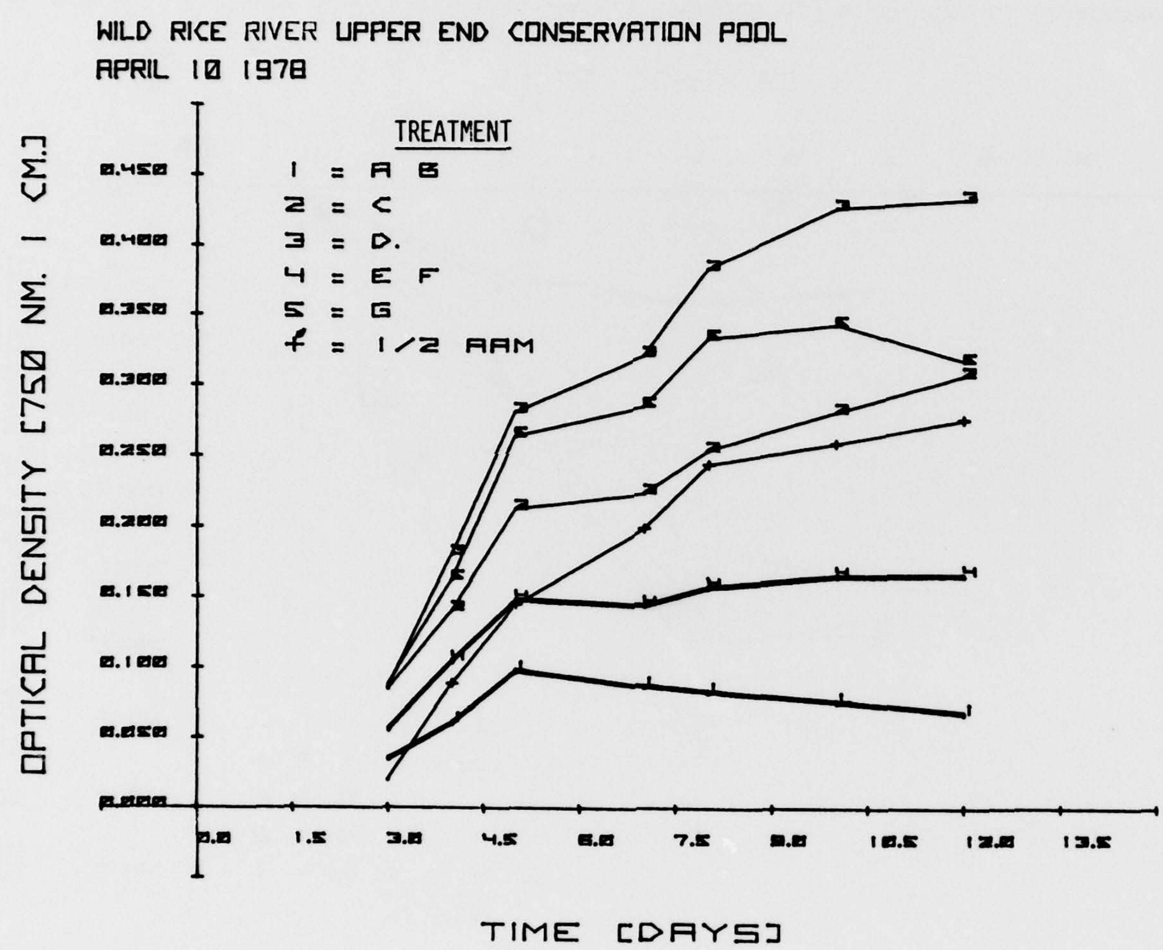


Figure A6. Optical density of sample 3 (April 10) with various treatments

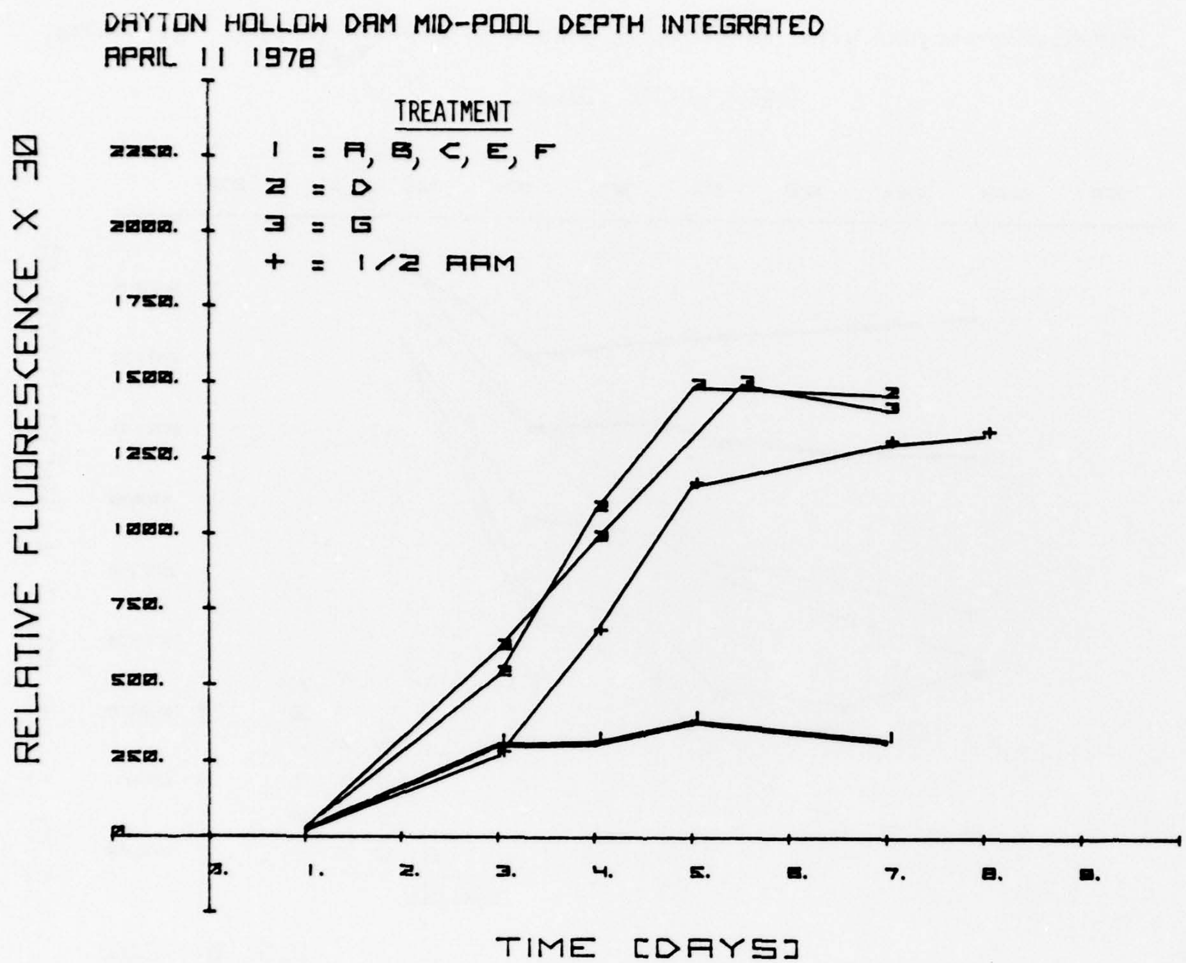


Figure A7. Relative fluorescence of sample 4 (April 11) with various treatments

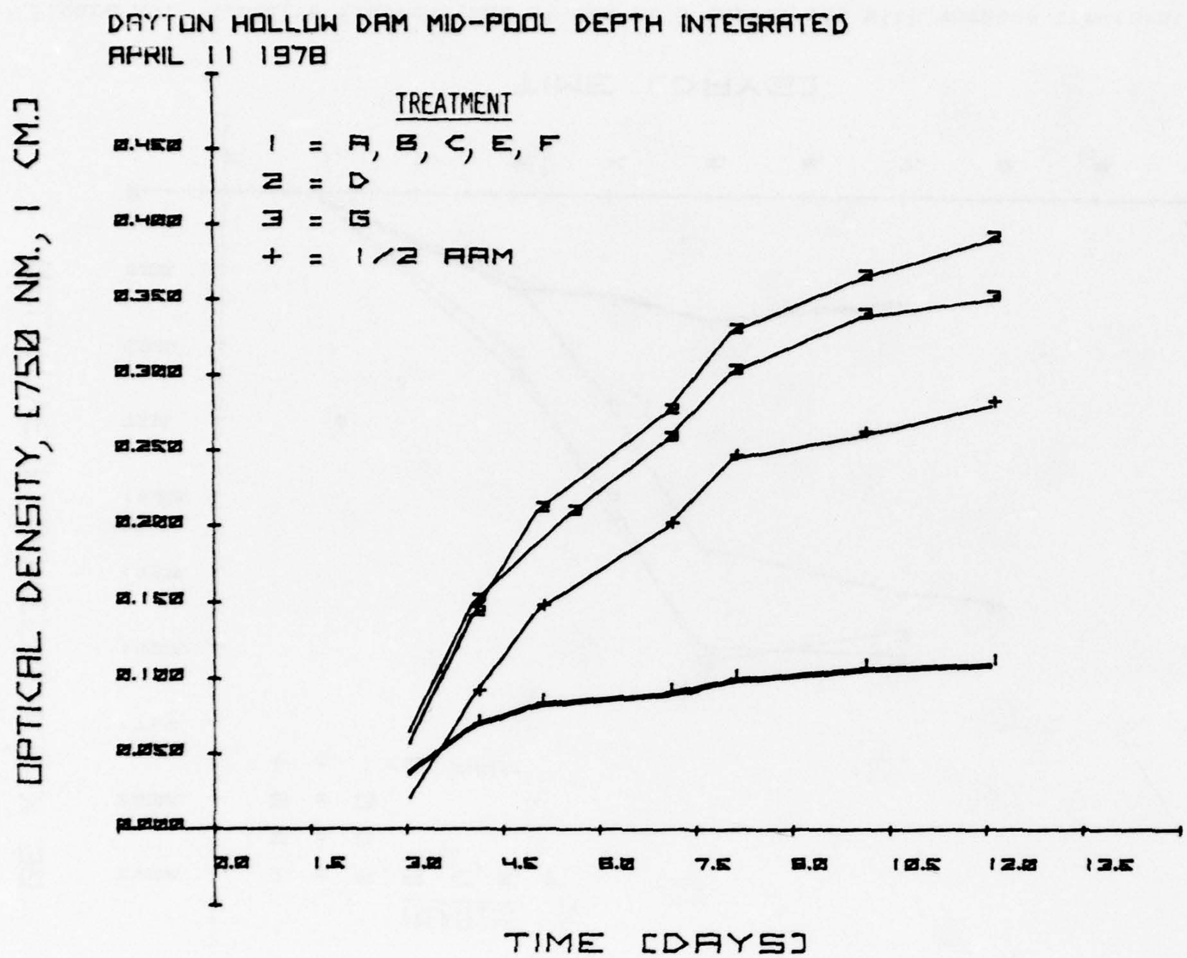


Figure A8. Optical density of sample 4 (April 11) with various treatments

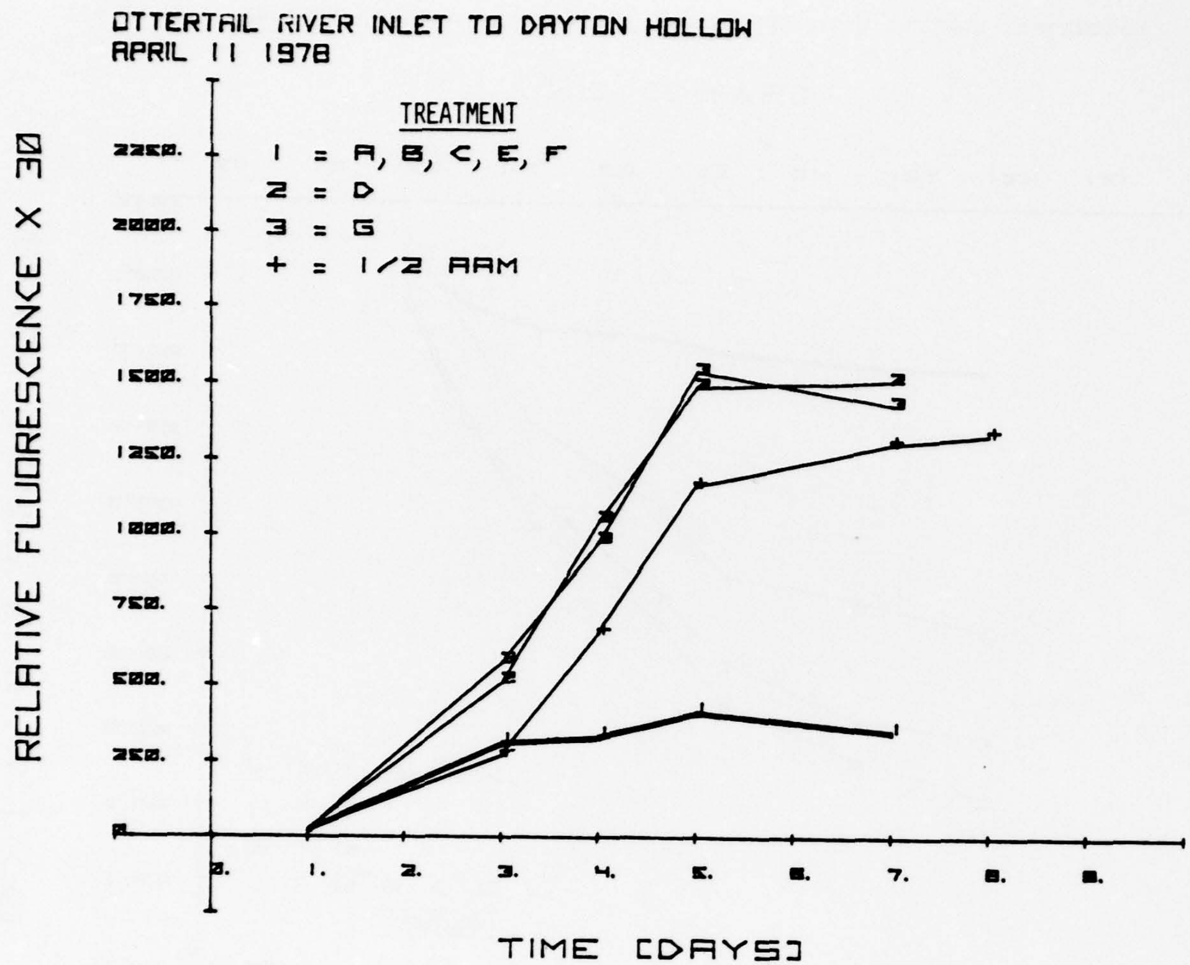


Figure A9. Relative fluorescence of sample 5 (April 11) with various treatments

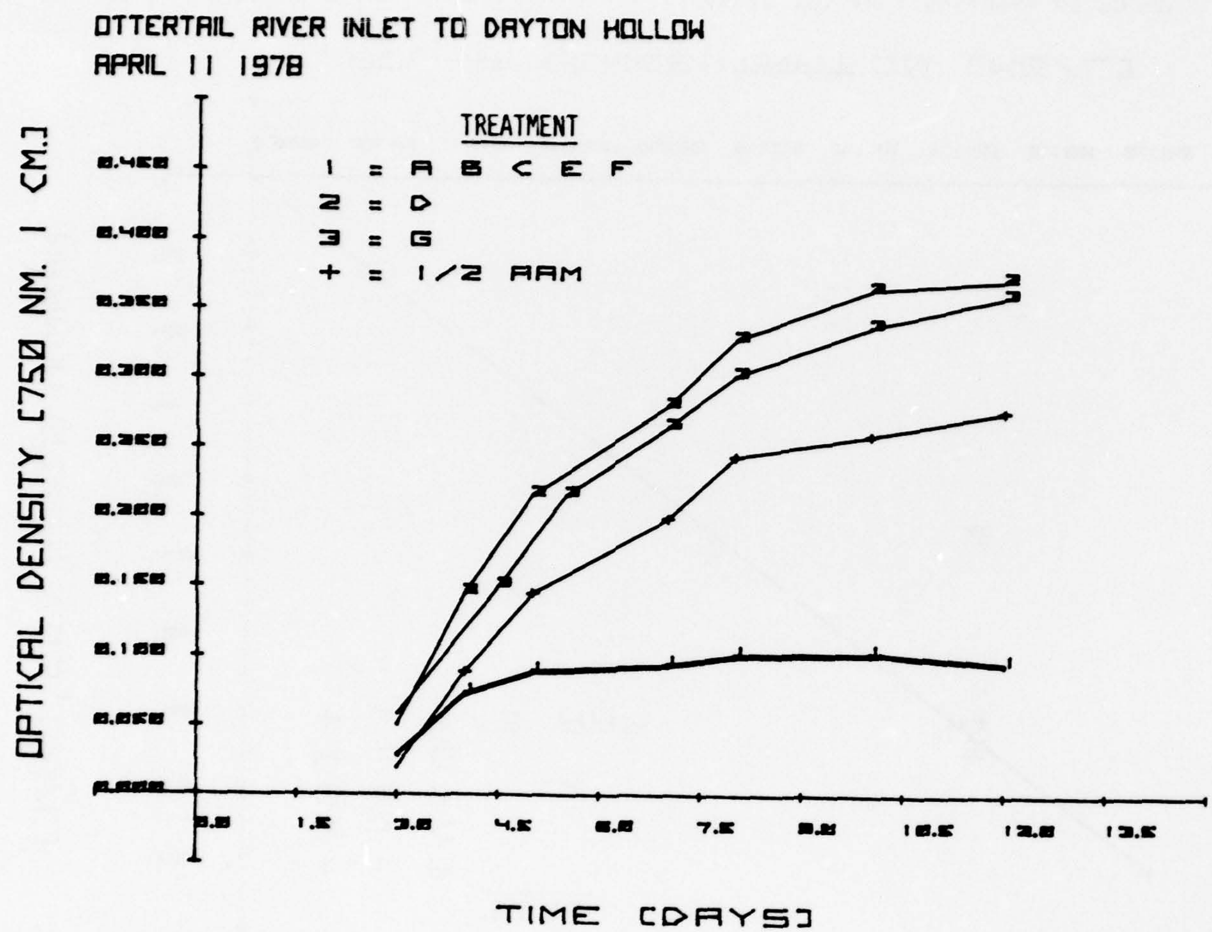


Figure A10. Optical density of sample 5 (April 11) with various treatments

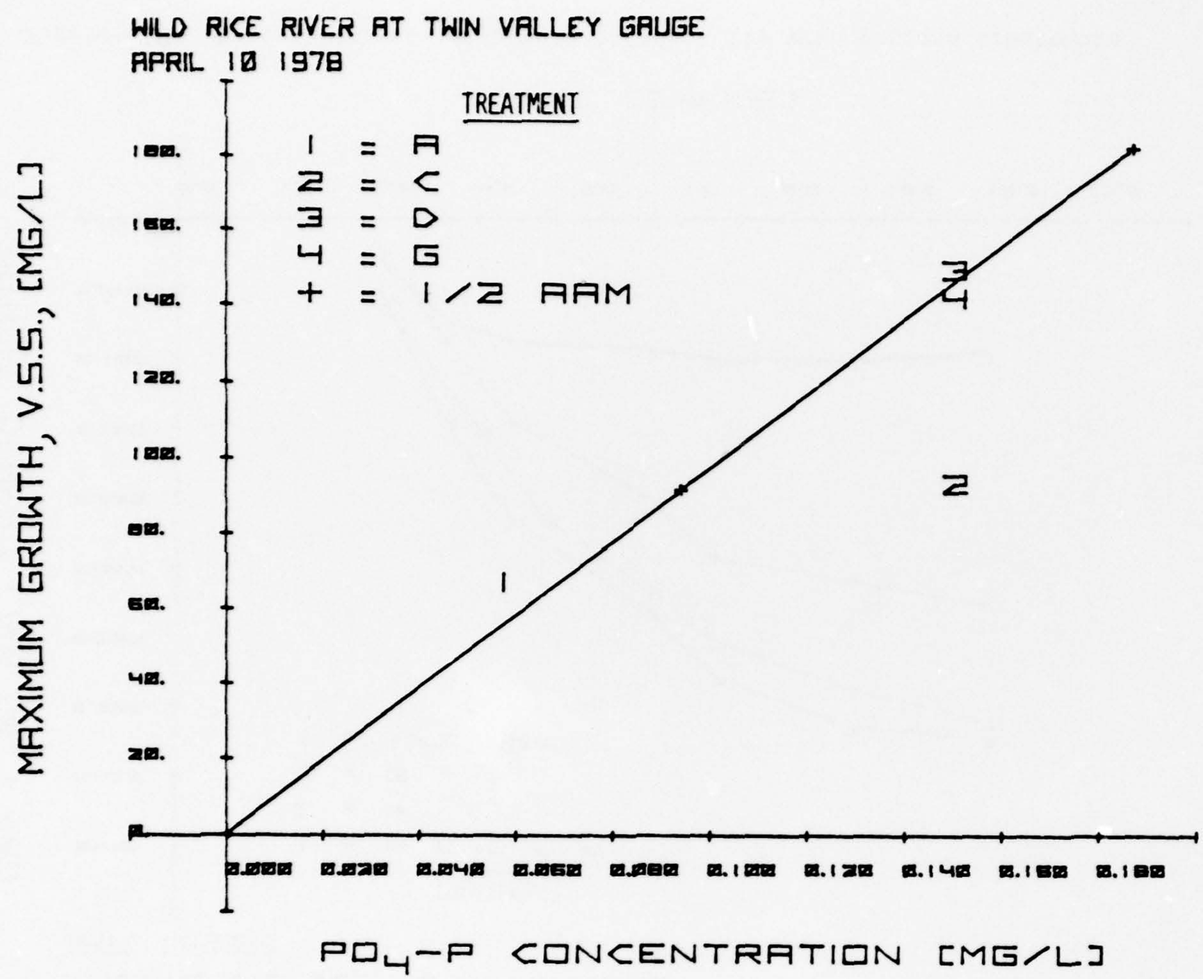


Figure A11. Maximum growth in sample 1 (April 10) as a function of PO_4-P

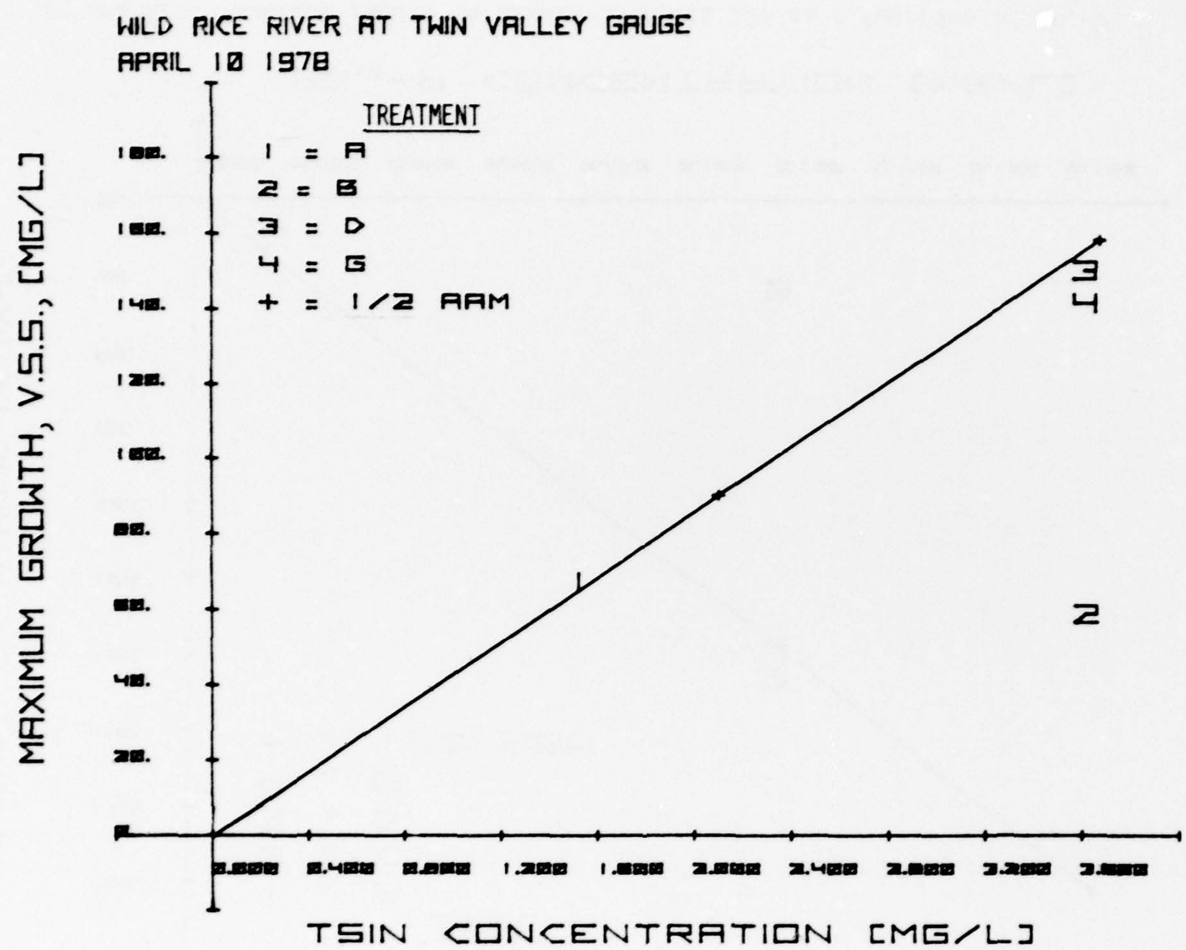


Figure A12. Maximum growth in sample 1 (April 10) as a function of TSIN

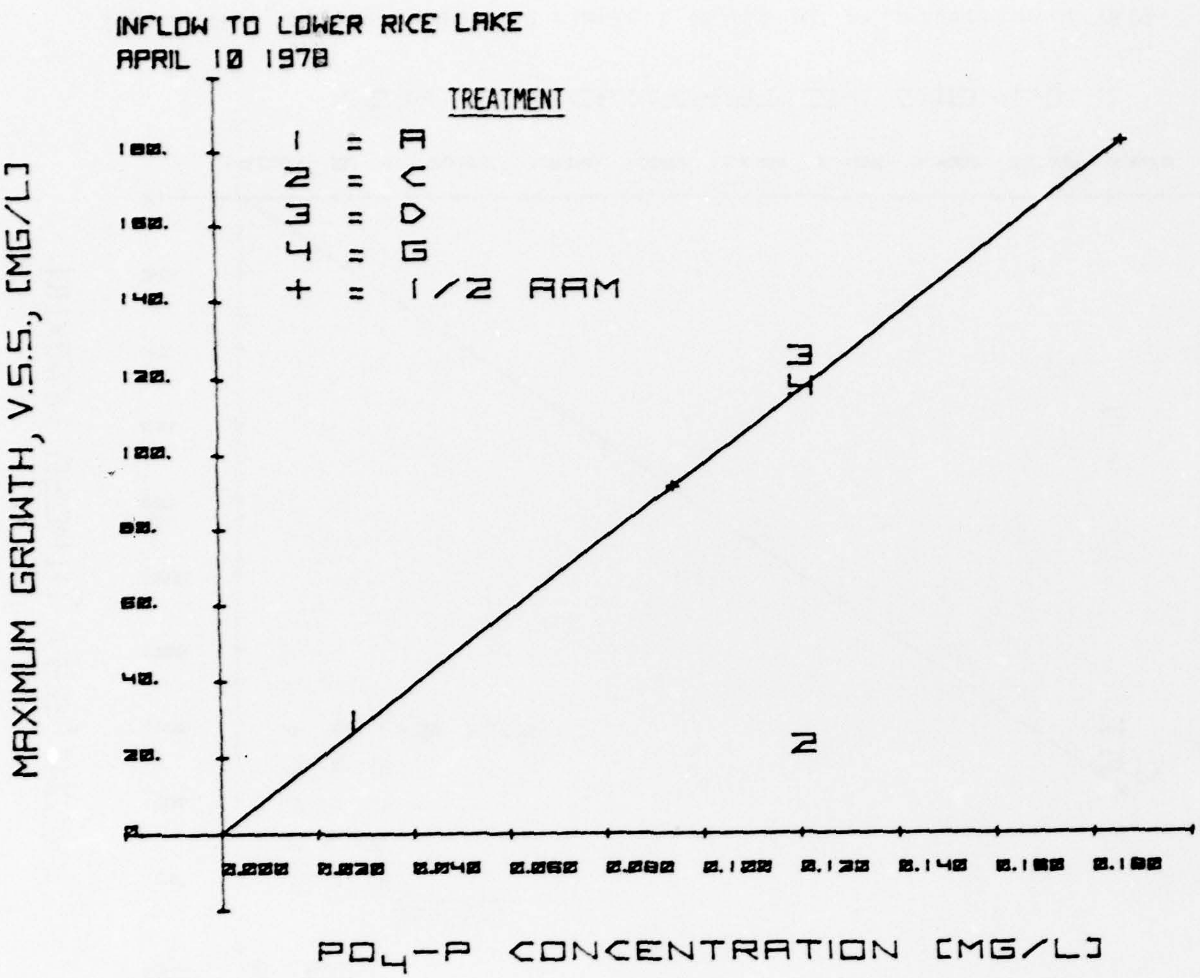


Figure A13. Maximum growth in sample 2 (April 10) as a function of PO_4--P

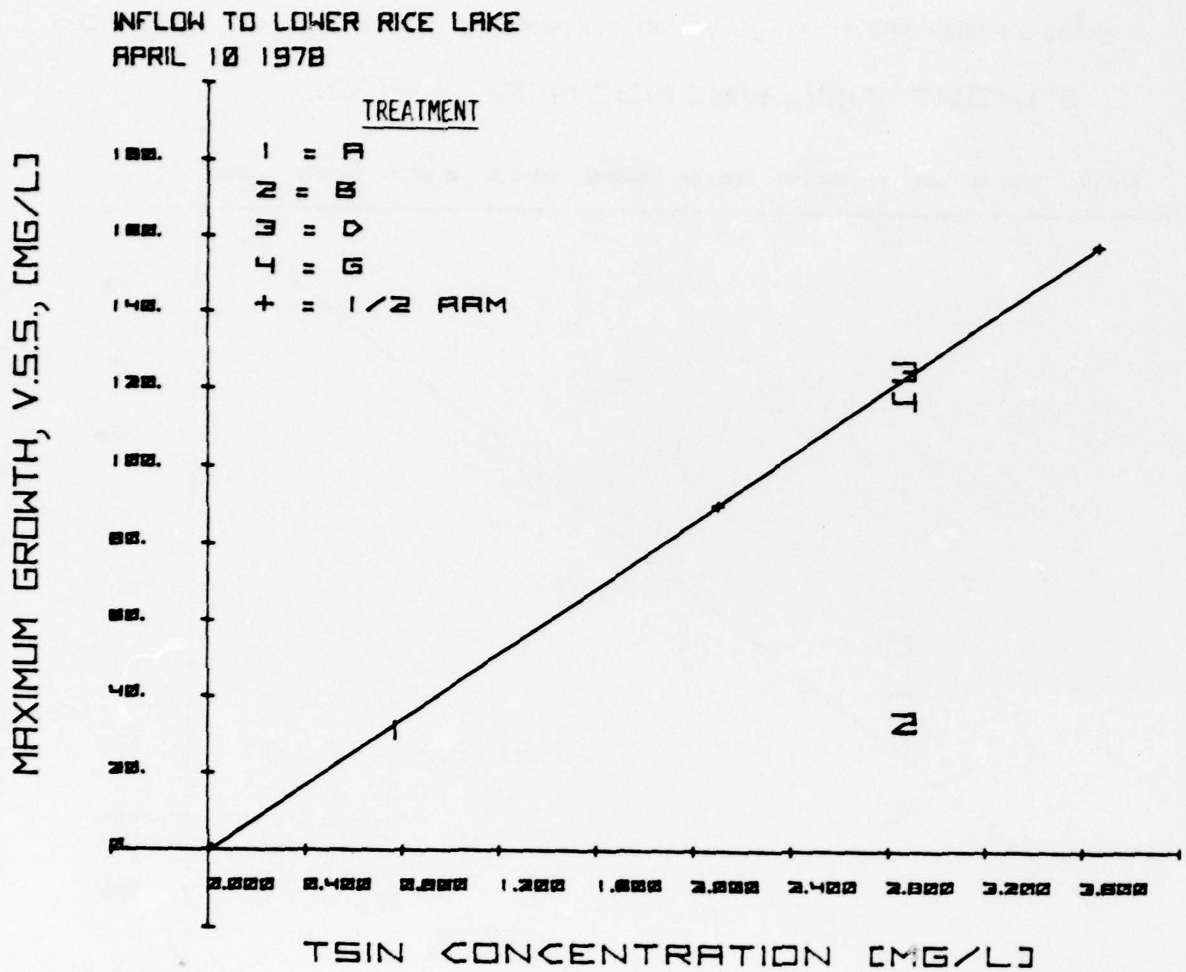


Figure A14. Maximum growth in sample 2 (April 10) as a function of TSIN

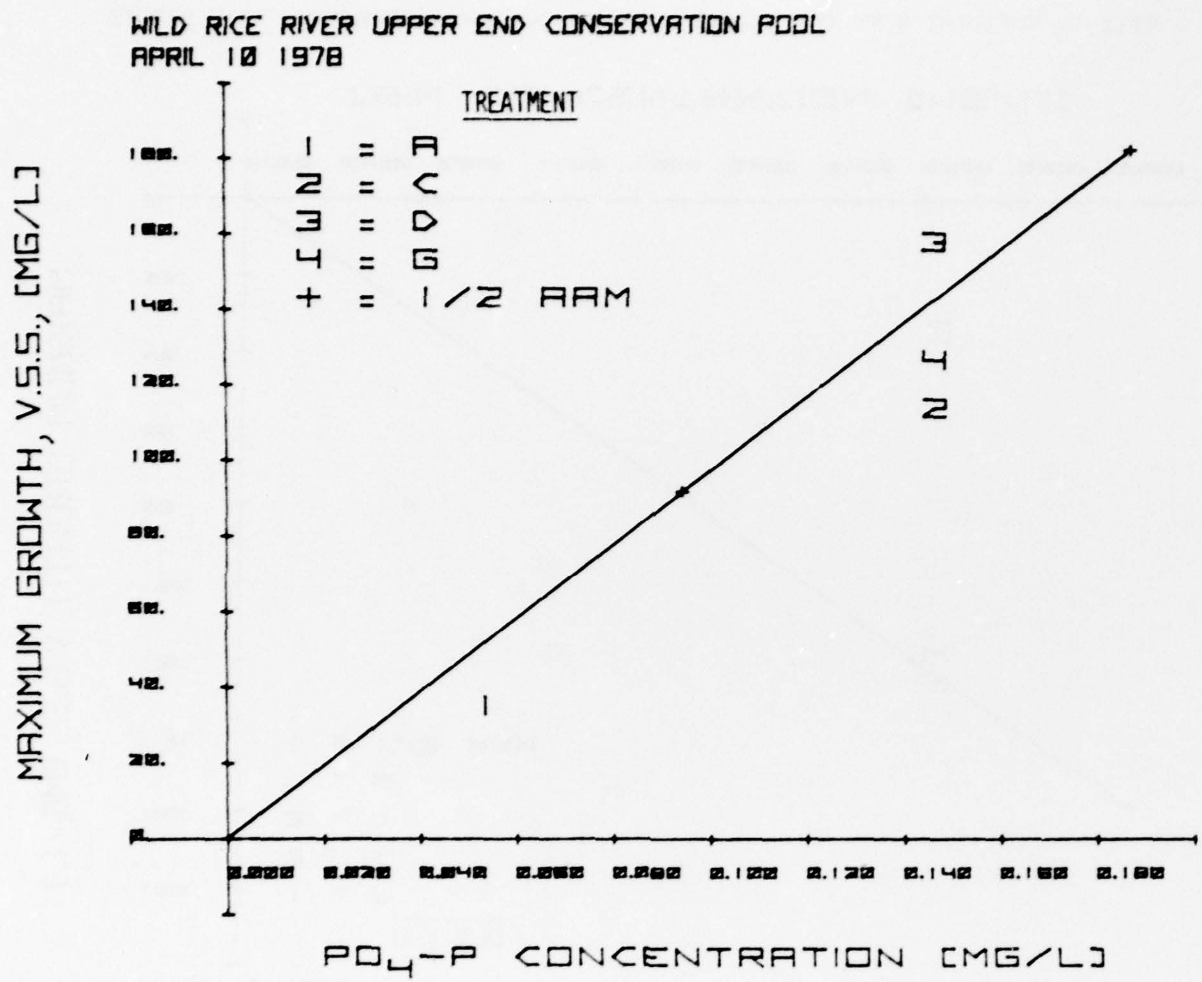


Figure A15. Maximum growth in sample 3 (April 10) as a function of PO_4-P

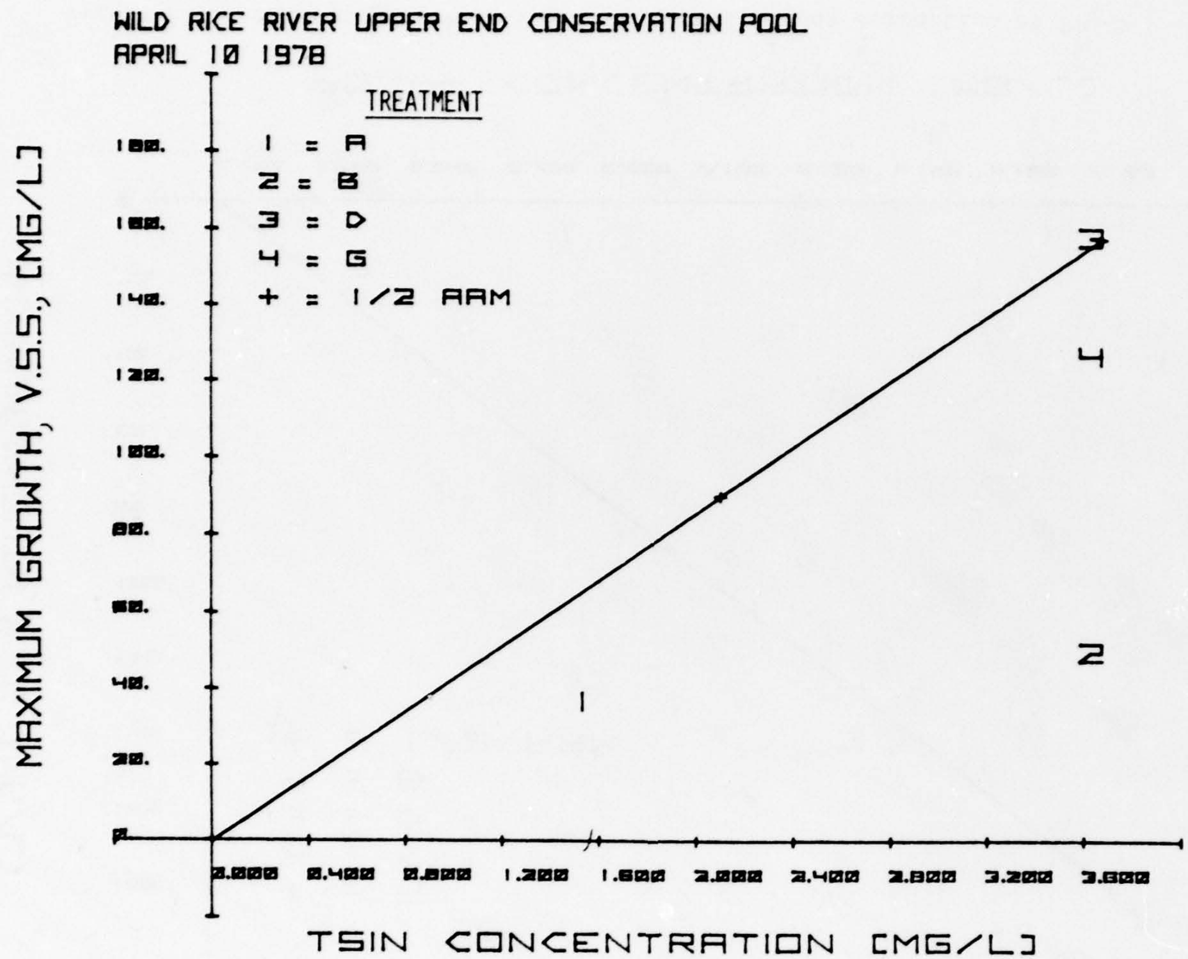


Figure A16. Maximum growth in sample 3 (April 10) as a function of TSIN

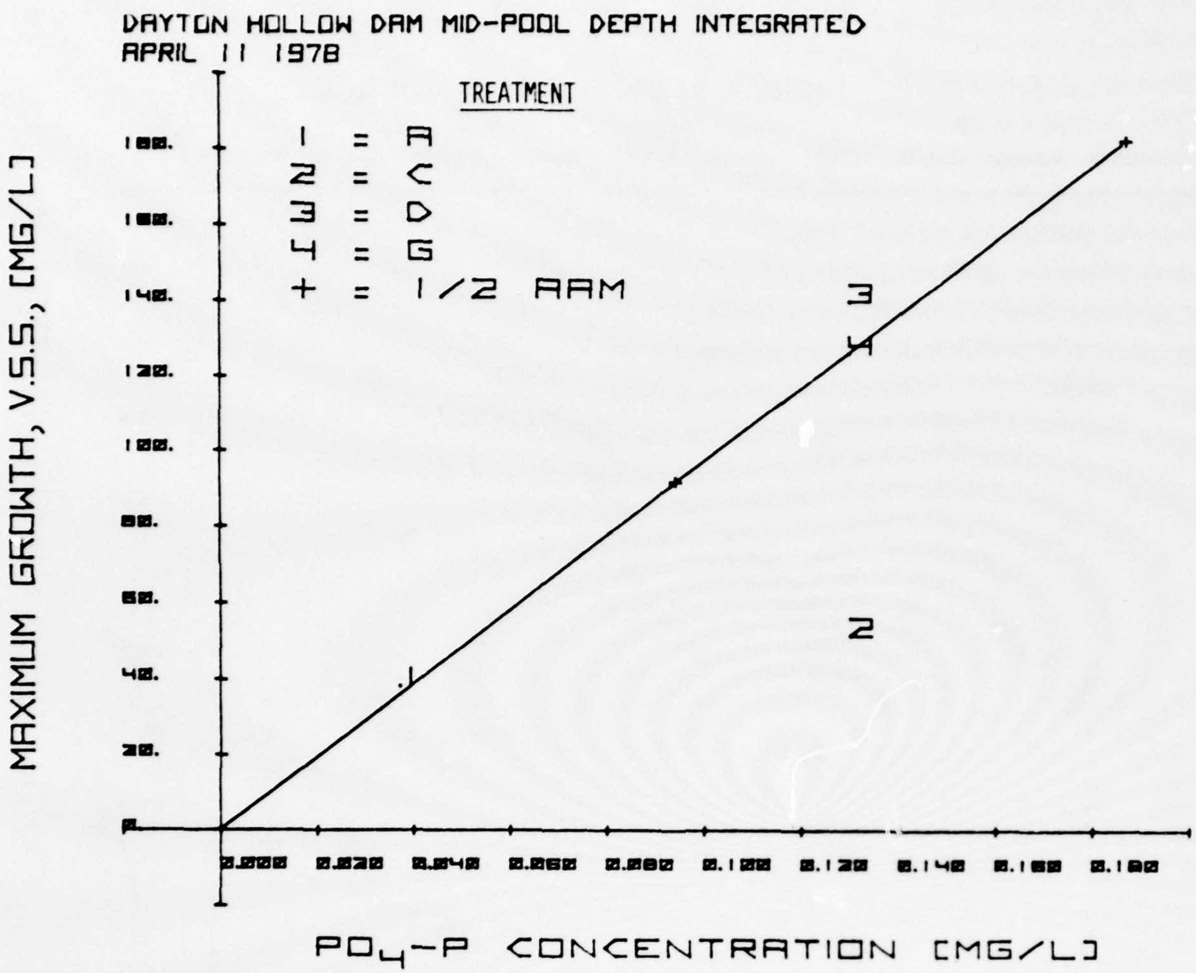


Figure A17. Maximum growth in sample 4 (April 11) as a function of PO_4-P

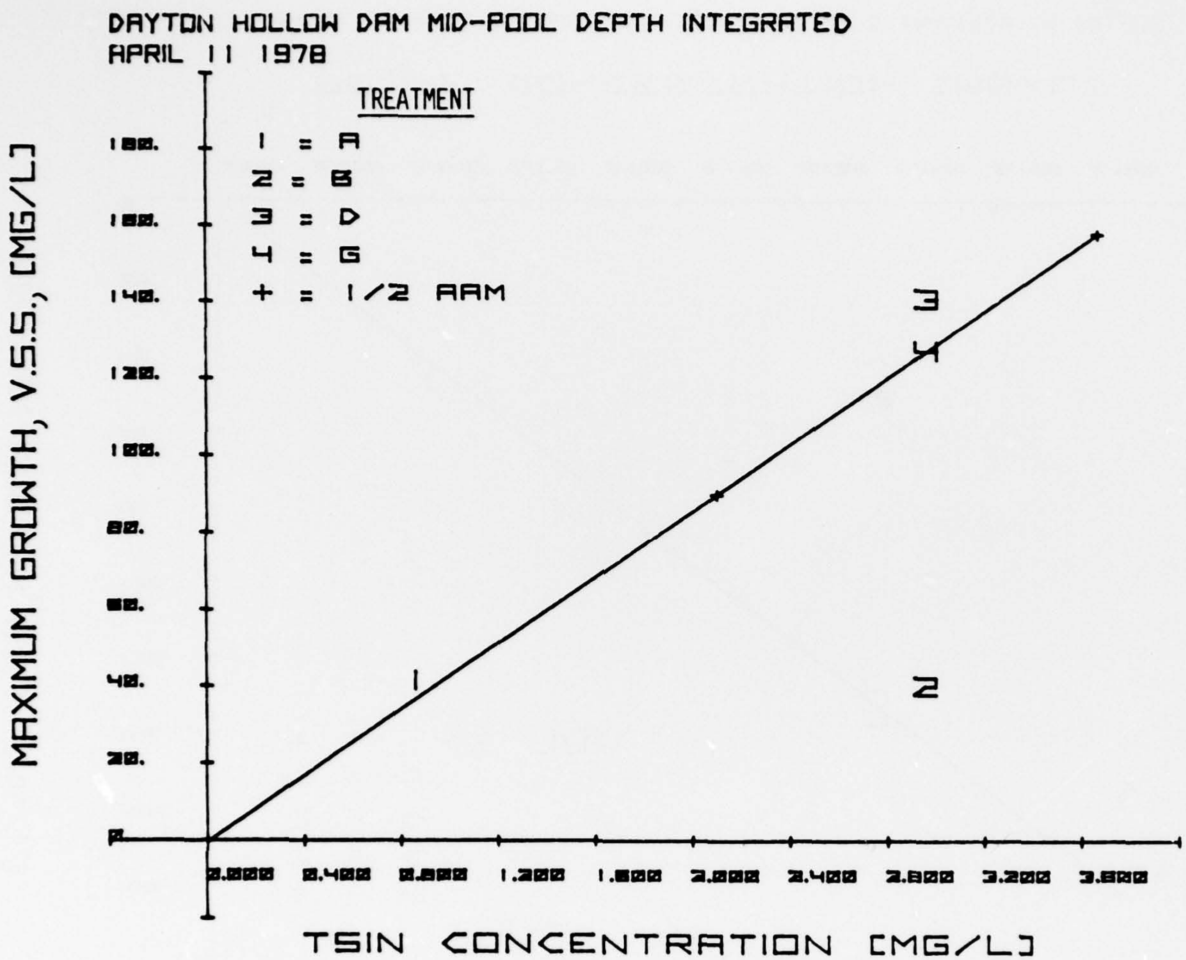


Figure A18. Maximum growth in sample 4 (April 11) as a function of TSIN

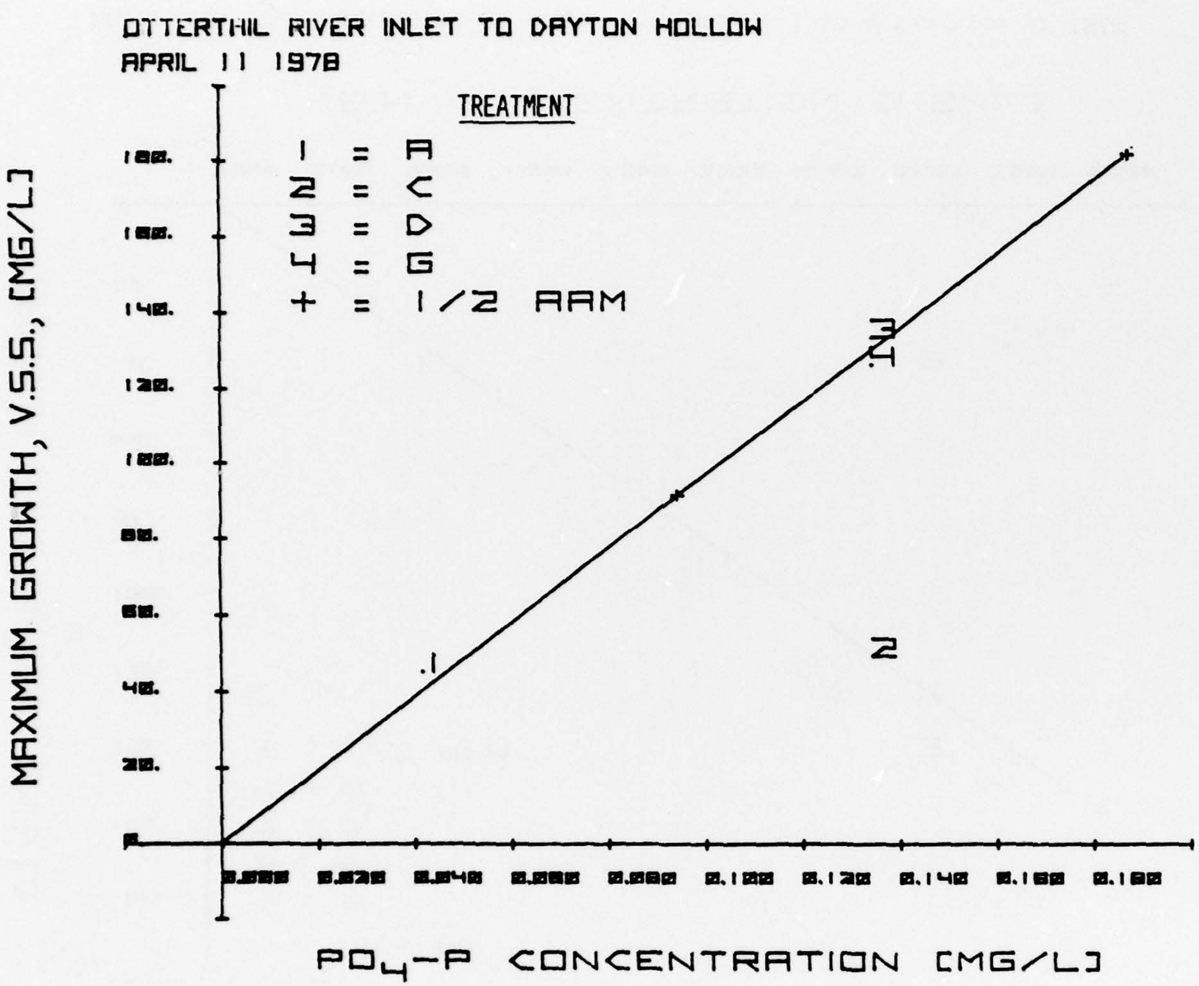


Figure A19. Maximum growth in sample 5 (April 11) as a function of PO_4-P

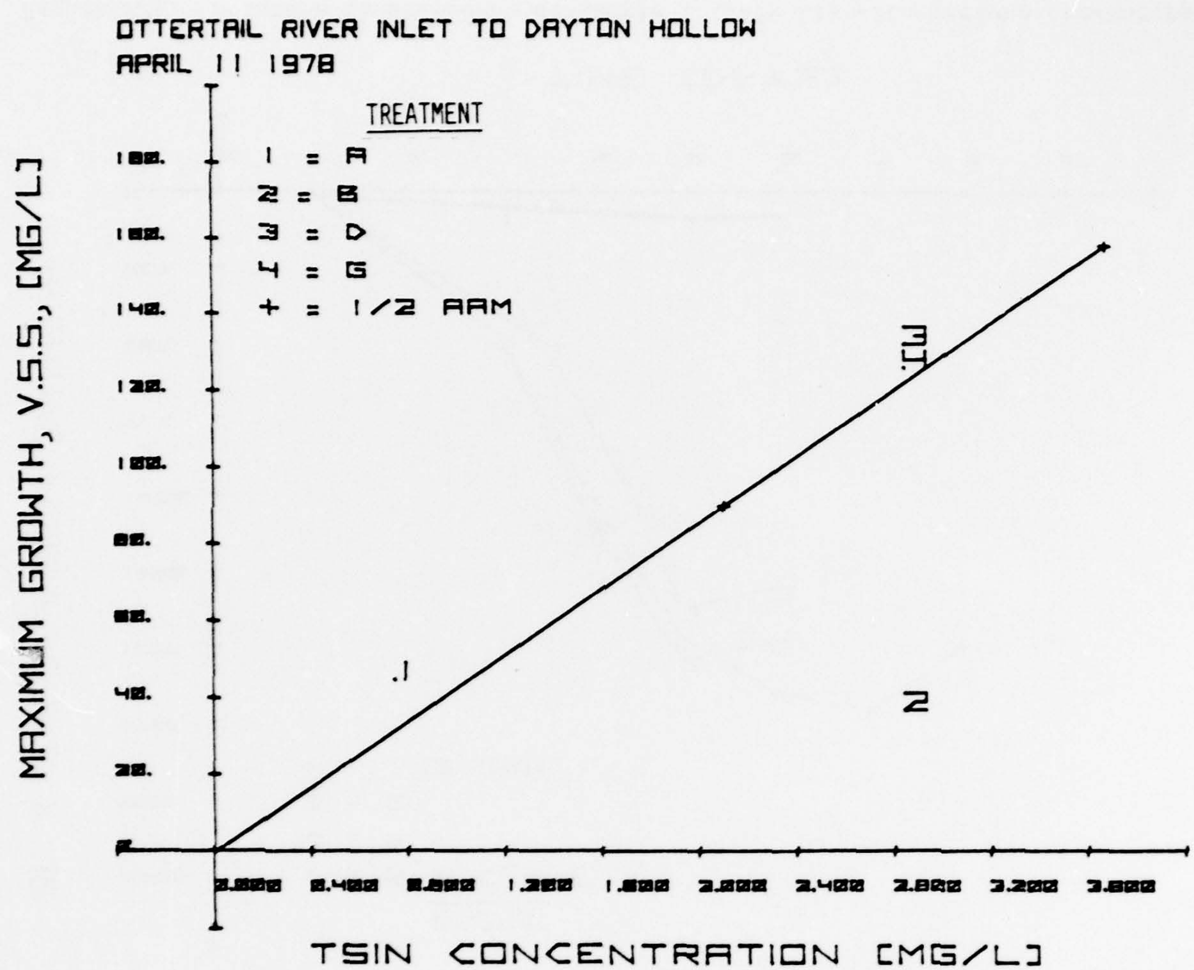


Figure A20. Maximum growth in sample 5 (April 11) as a function of TSIN

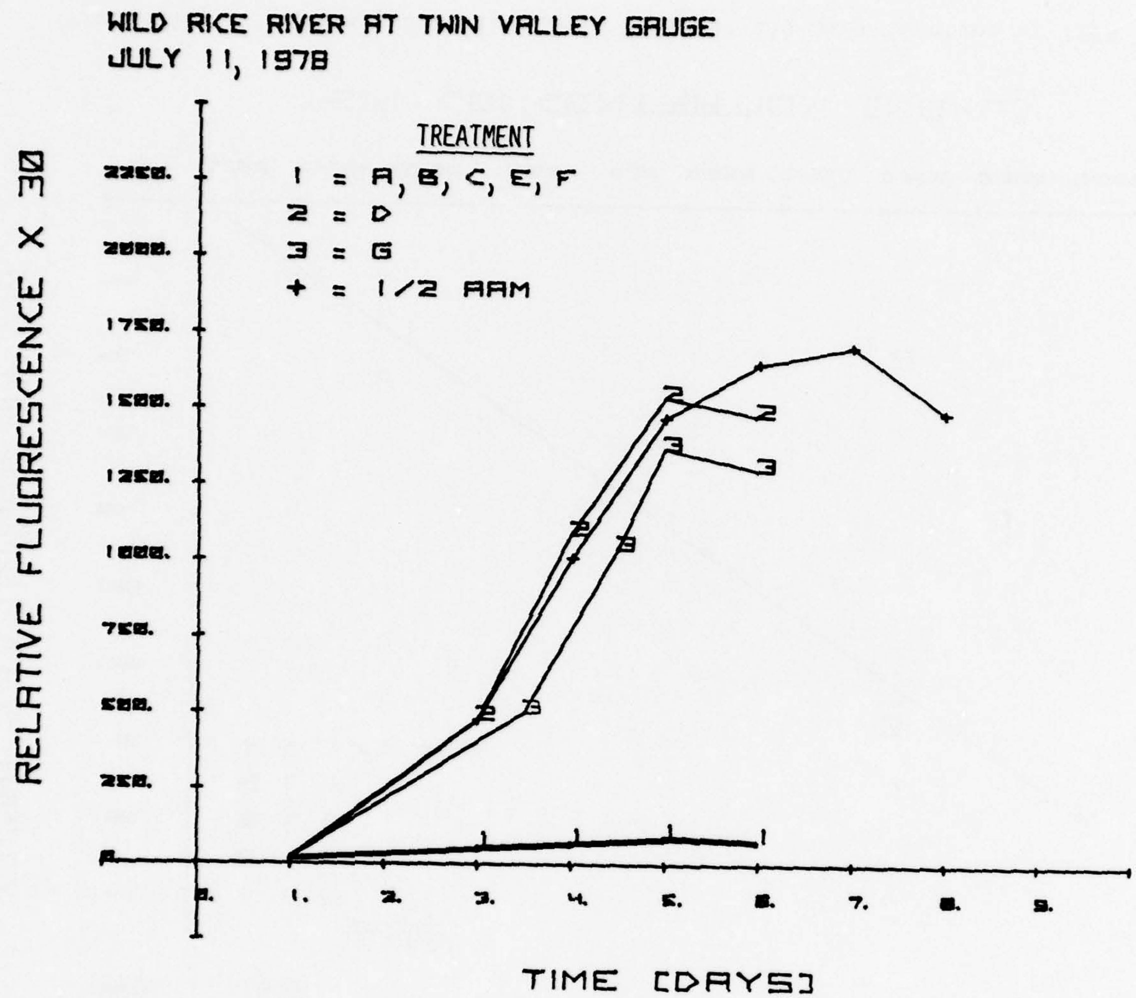


Figure A21. Relative fluorescence of sample 1 (July 11) with various treatments

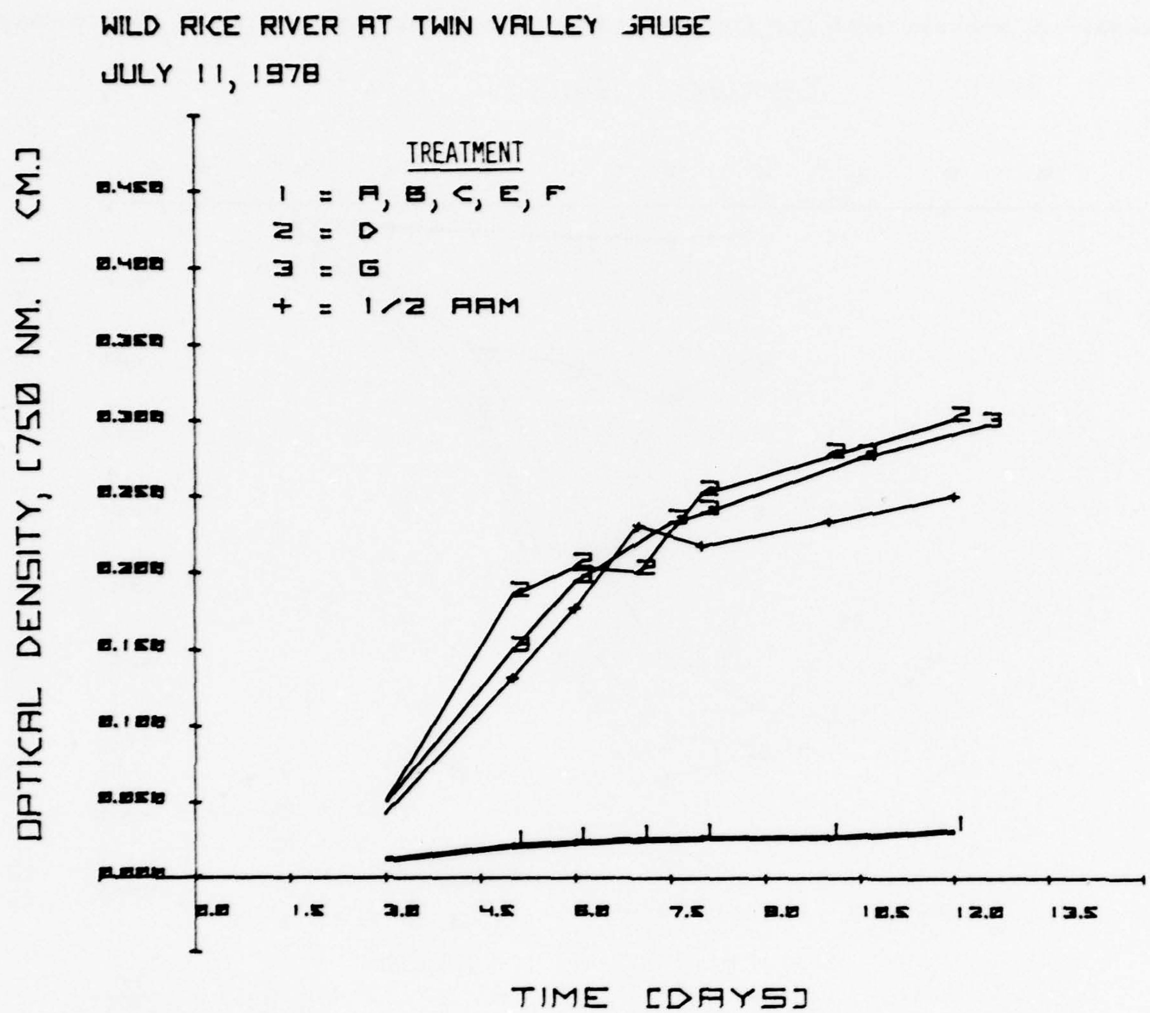


Figure A22. Optical density of sample 1 (July 11) with various treatments

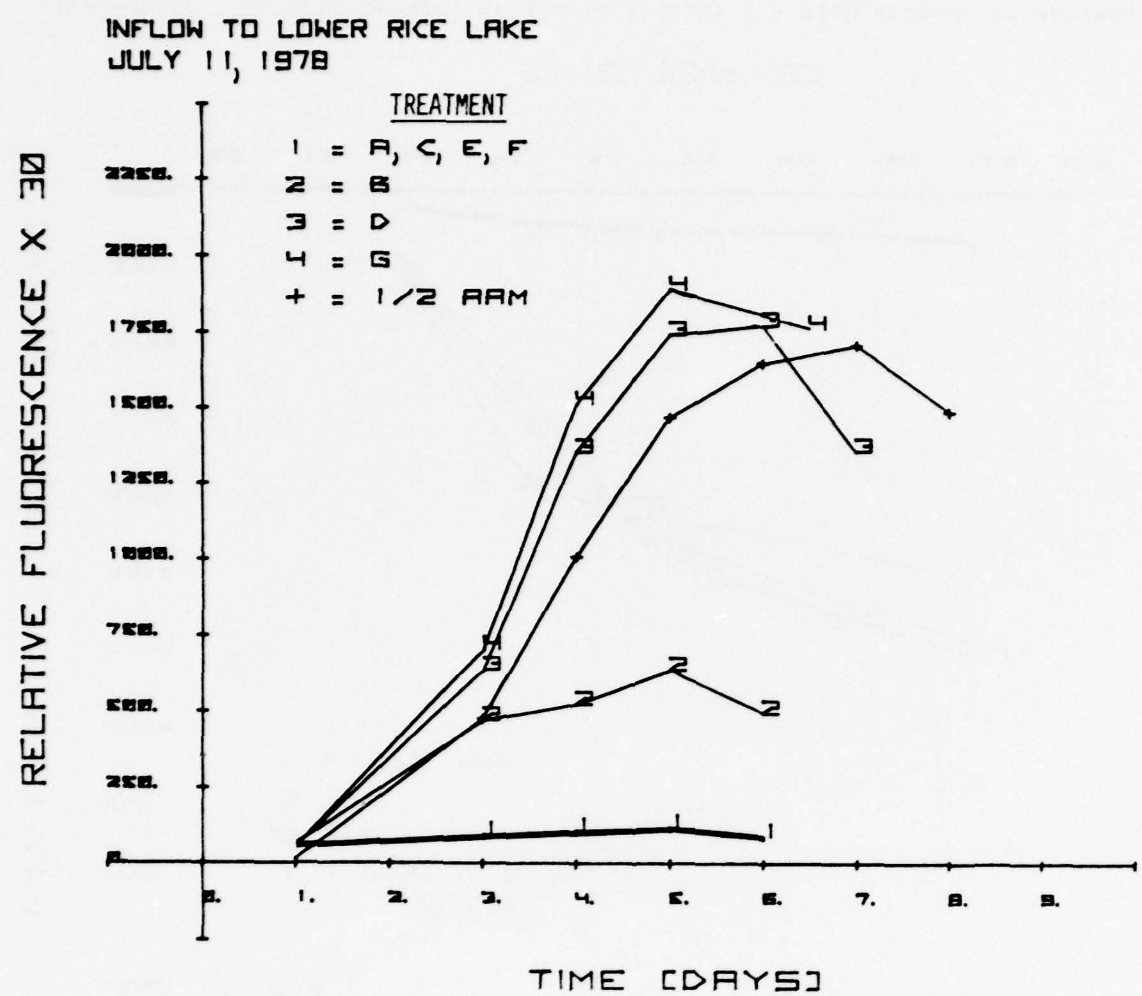


Figure A23. Relative fluorescence of sample 2 (July 11) with various treatments

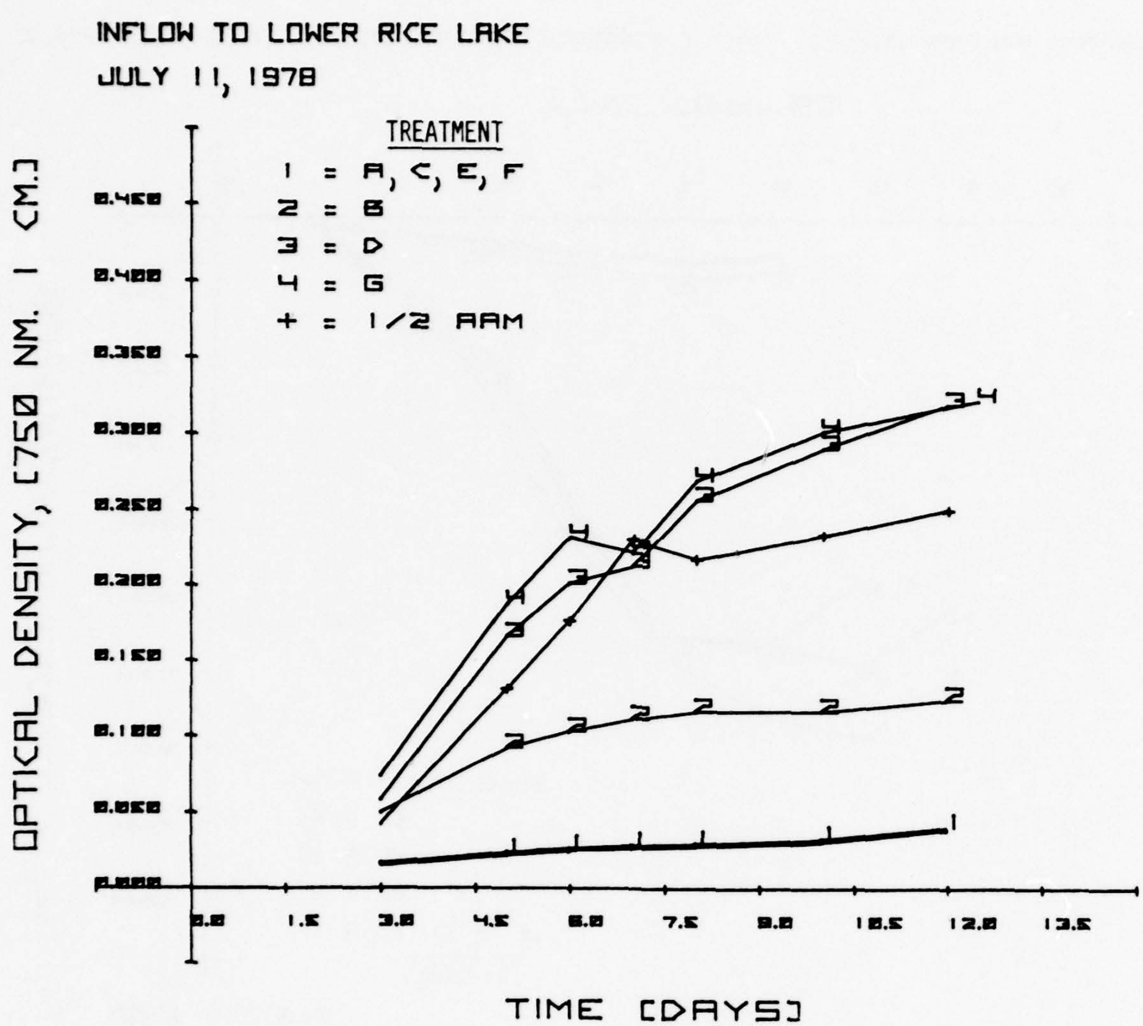


Figure A24. Optical density of sample 2 (July 11) with various treatments

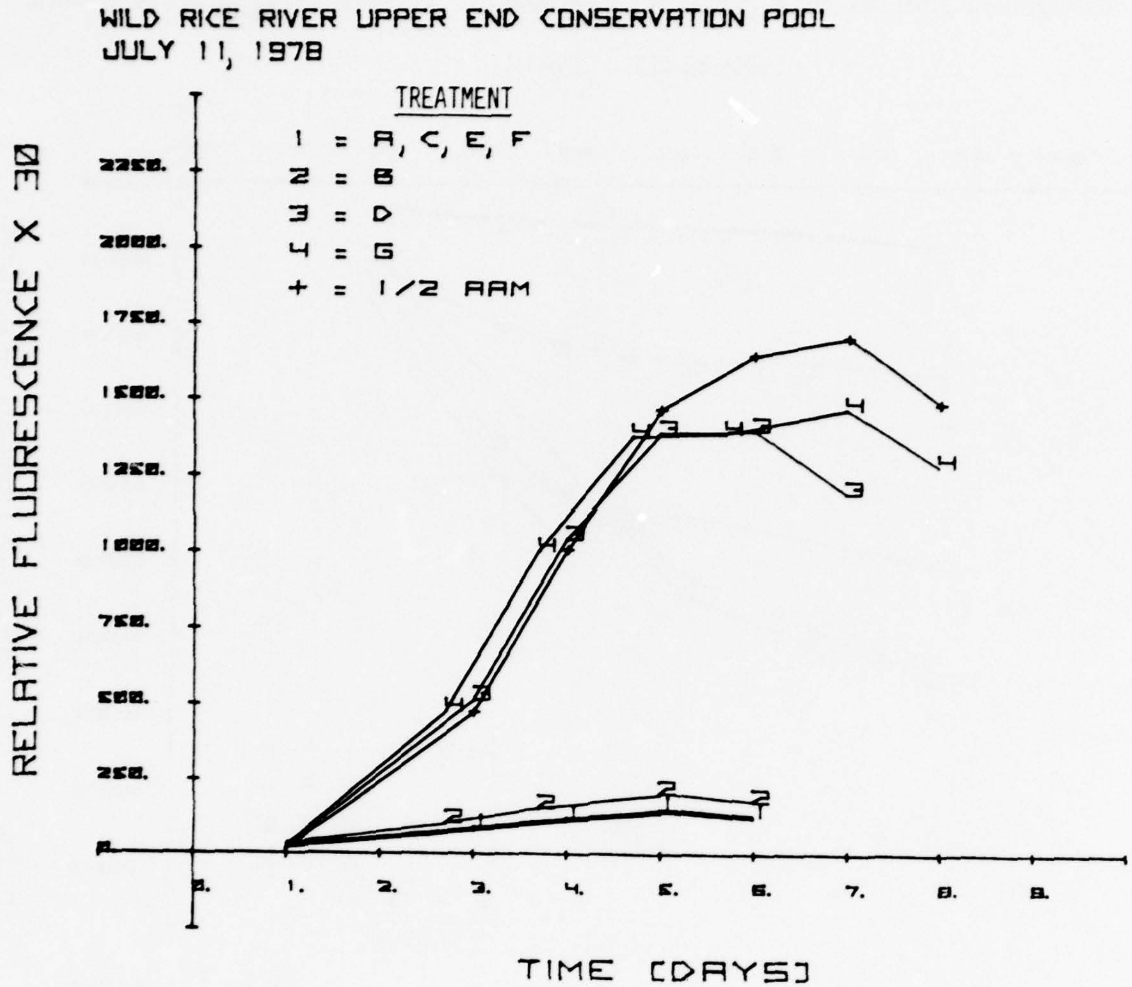


Figure A25. Relative fluorescence of sample 3 (July 11) with various treatments

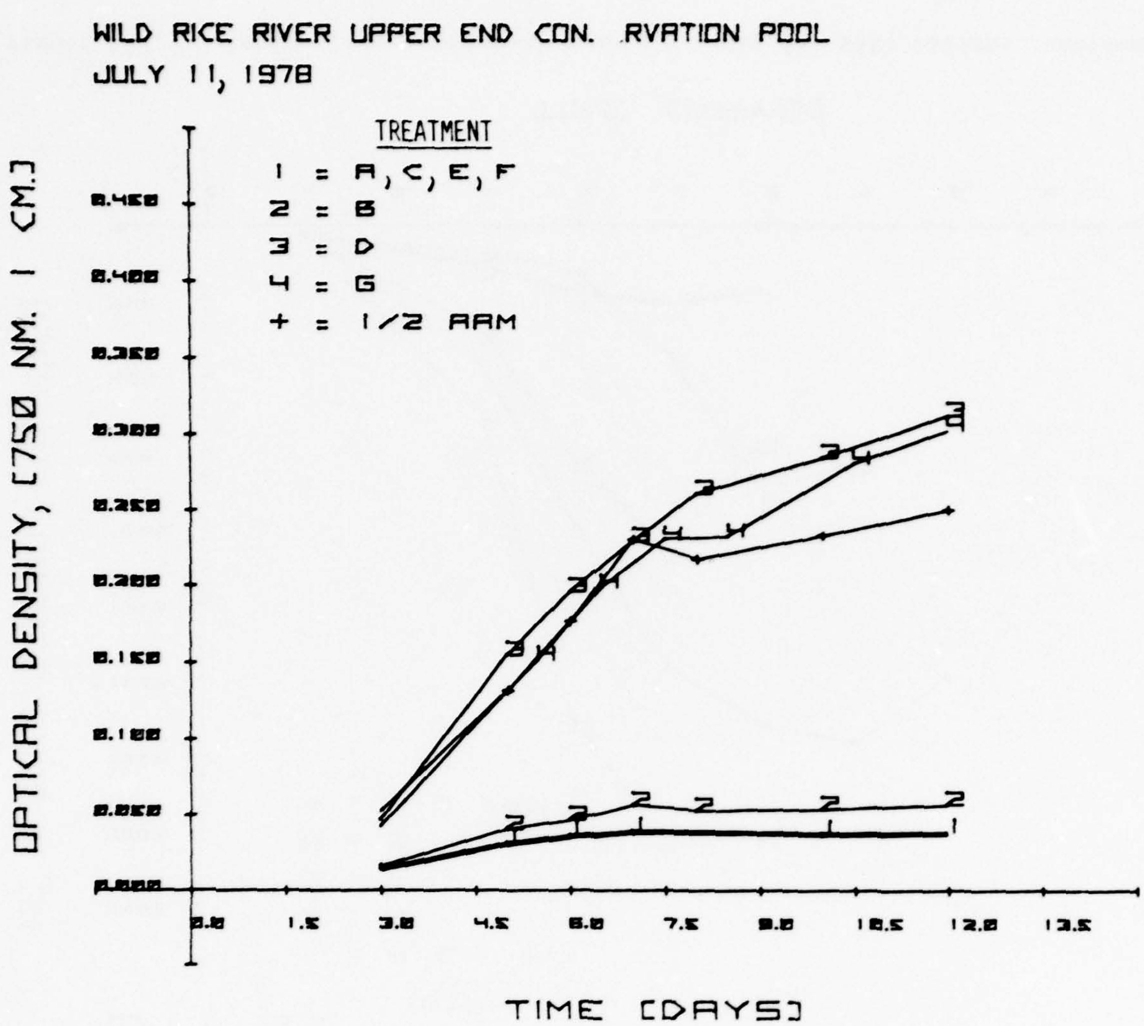


Figure A26. Optical density of sample 3 (July 11) with various treatments

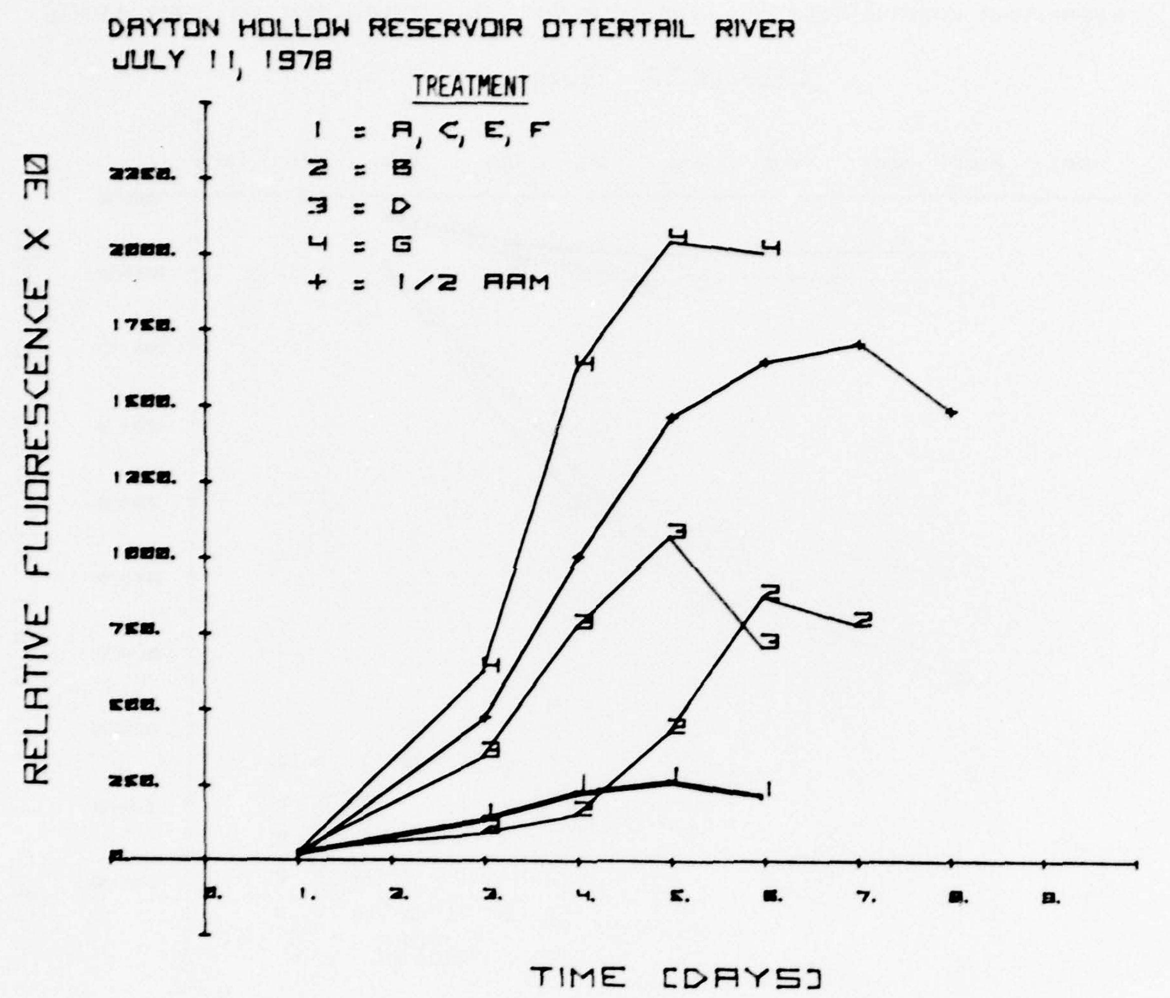


Figure A27. Relative fluorescence of sample 4 (July 11) with various treatments

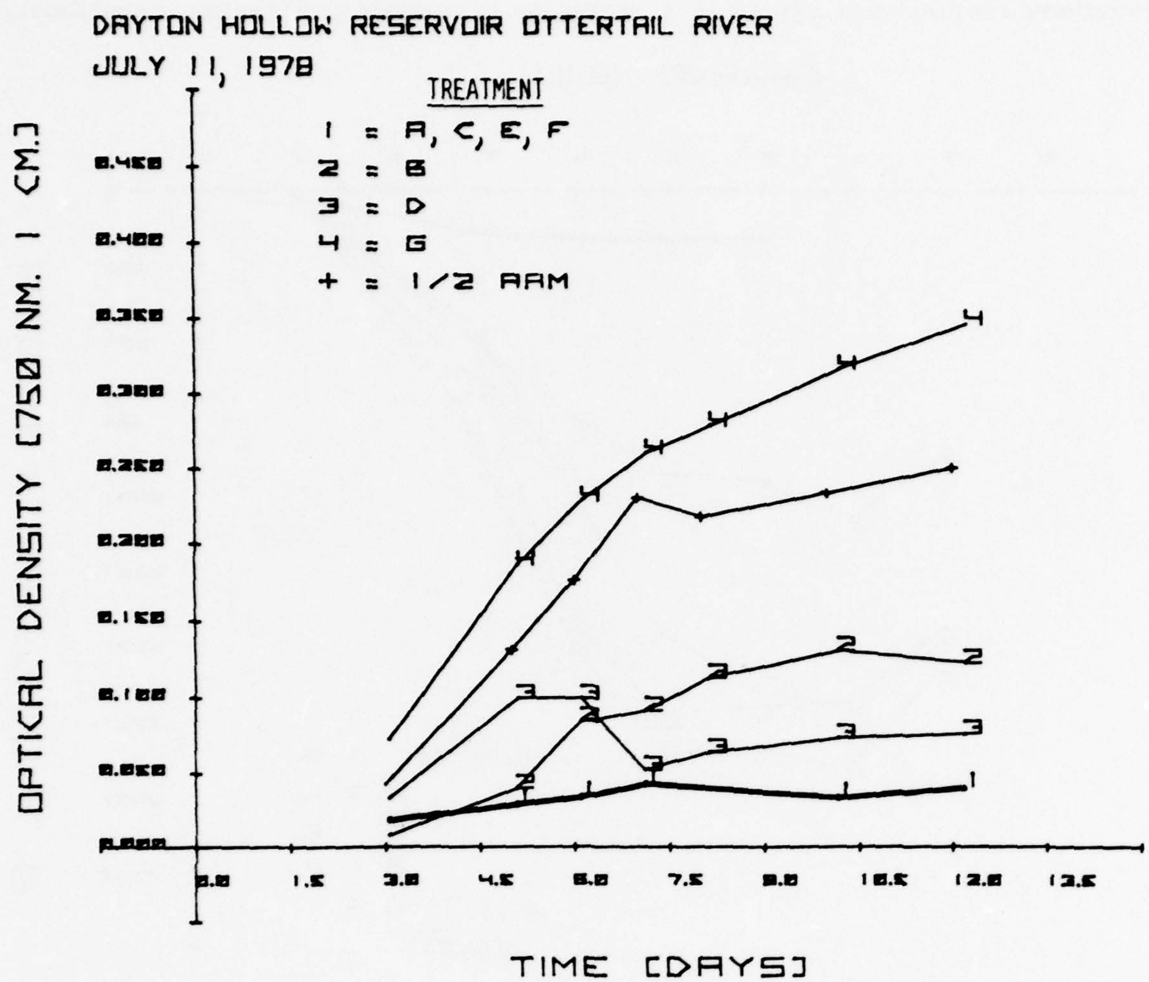


Figure A28. Optical density of sample 4 (July 11) with various treatments

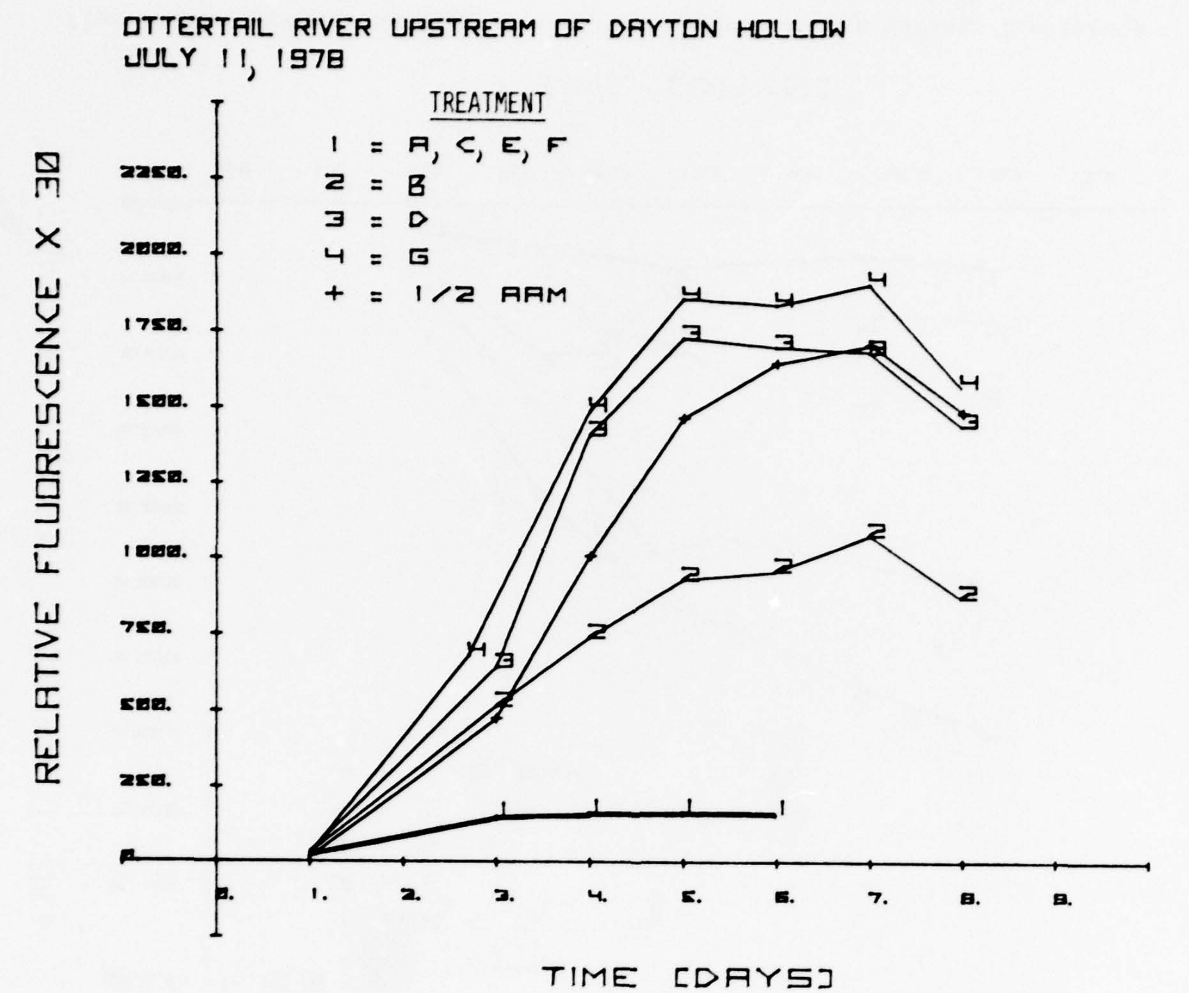


Figure A29. Relative fluorescence of sample 5 (July 11) with various treatments

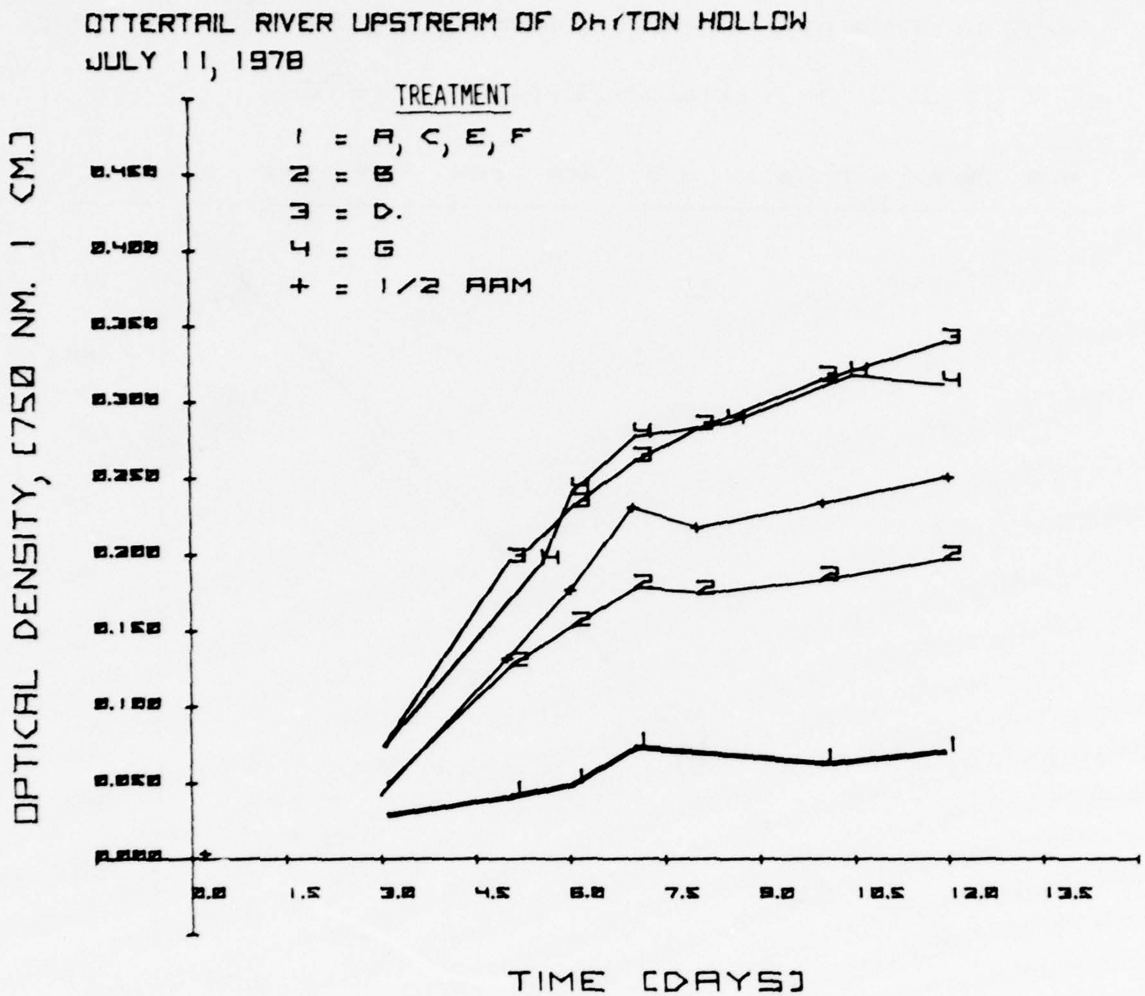


Figure A30. Optical density of sample 5 (July 11) with various treatments

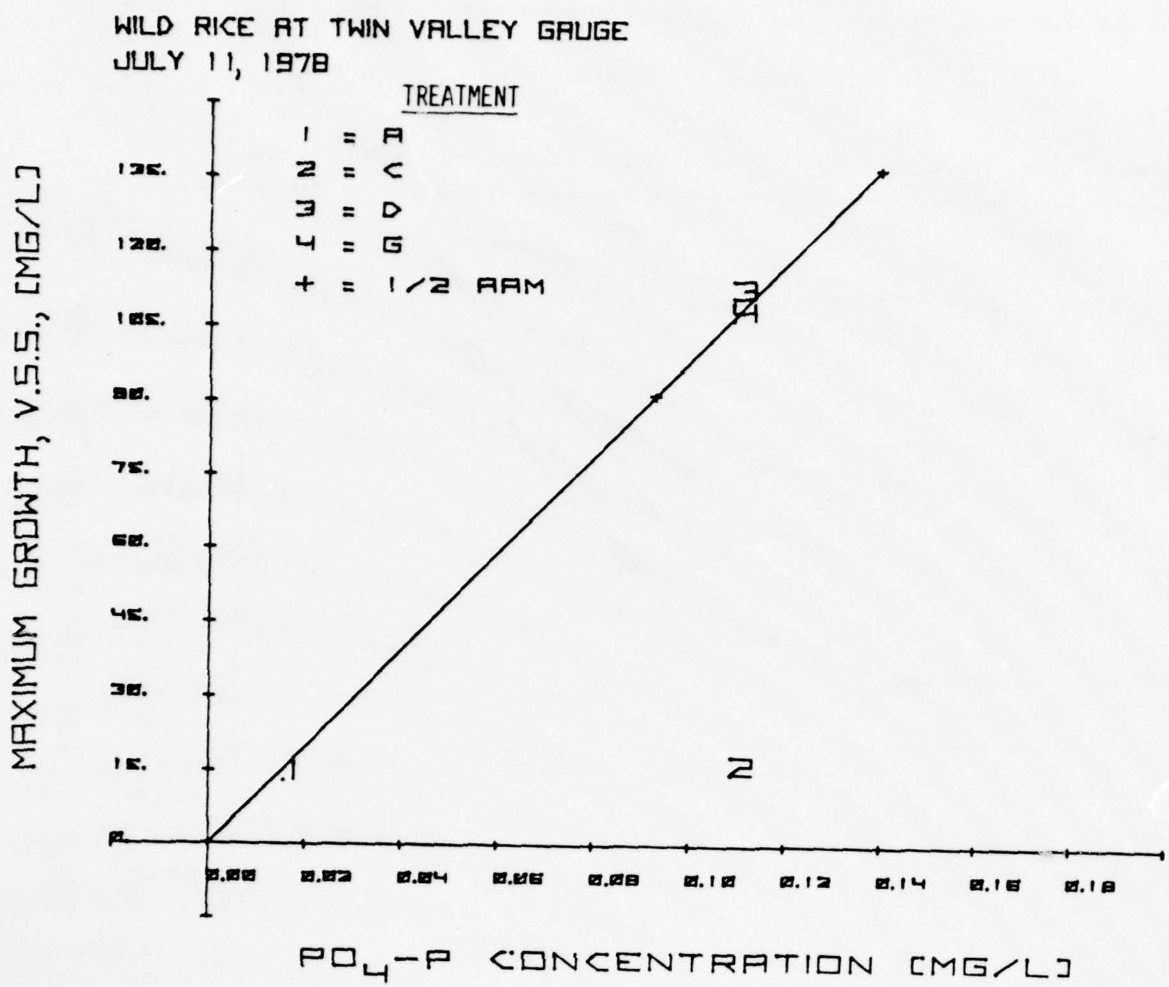


Figure A31. Maximum growth in sample 1 (July 11) as a function of PO_4-P

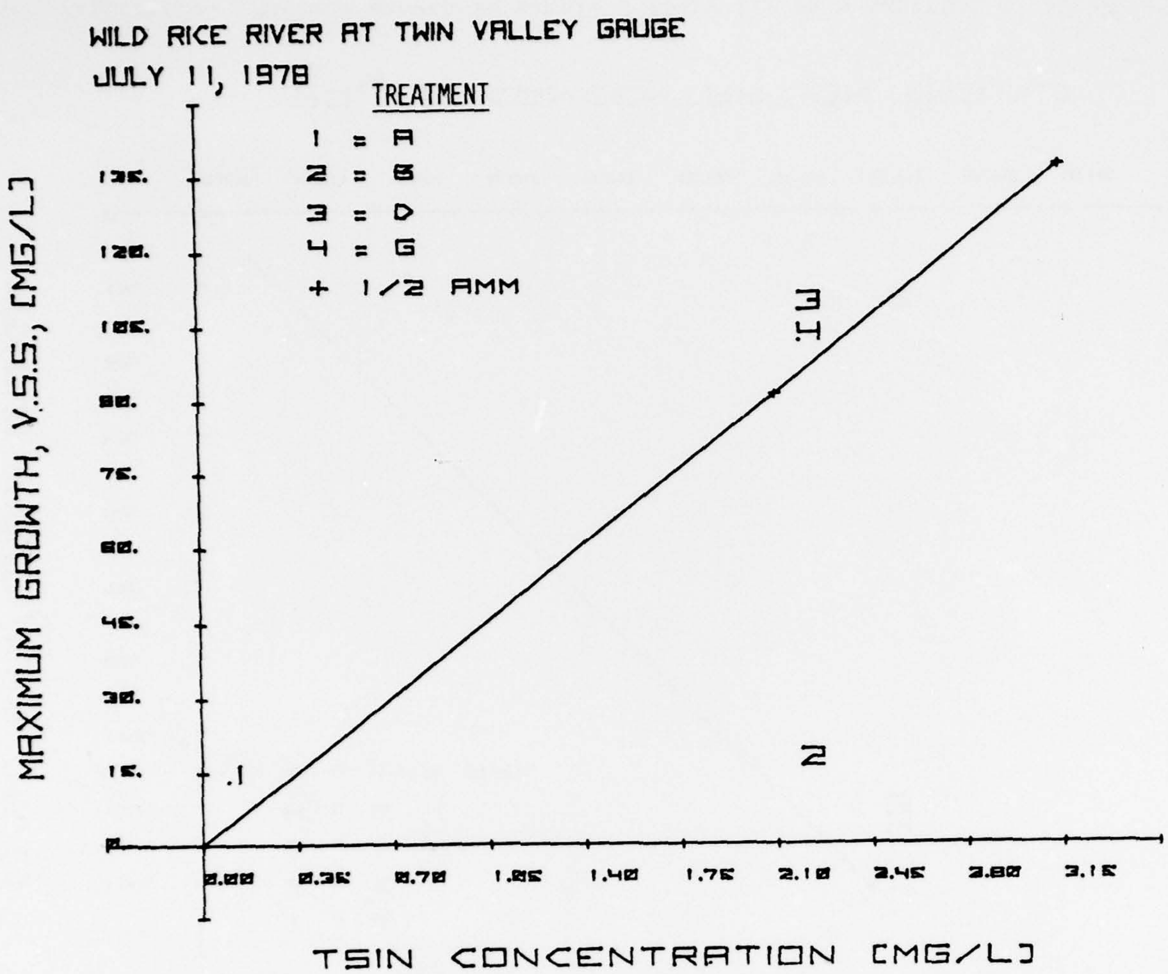


Figure A32. Maximum growth in sample 1 (July 11) as a function of TSIN

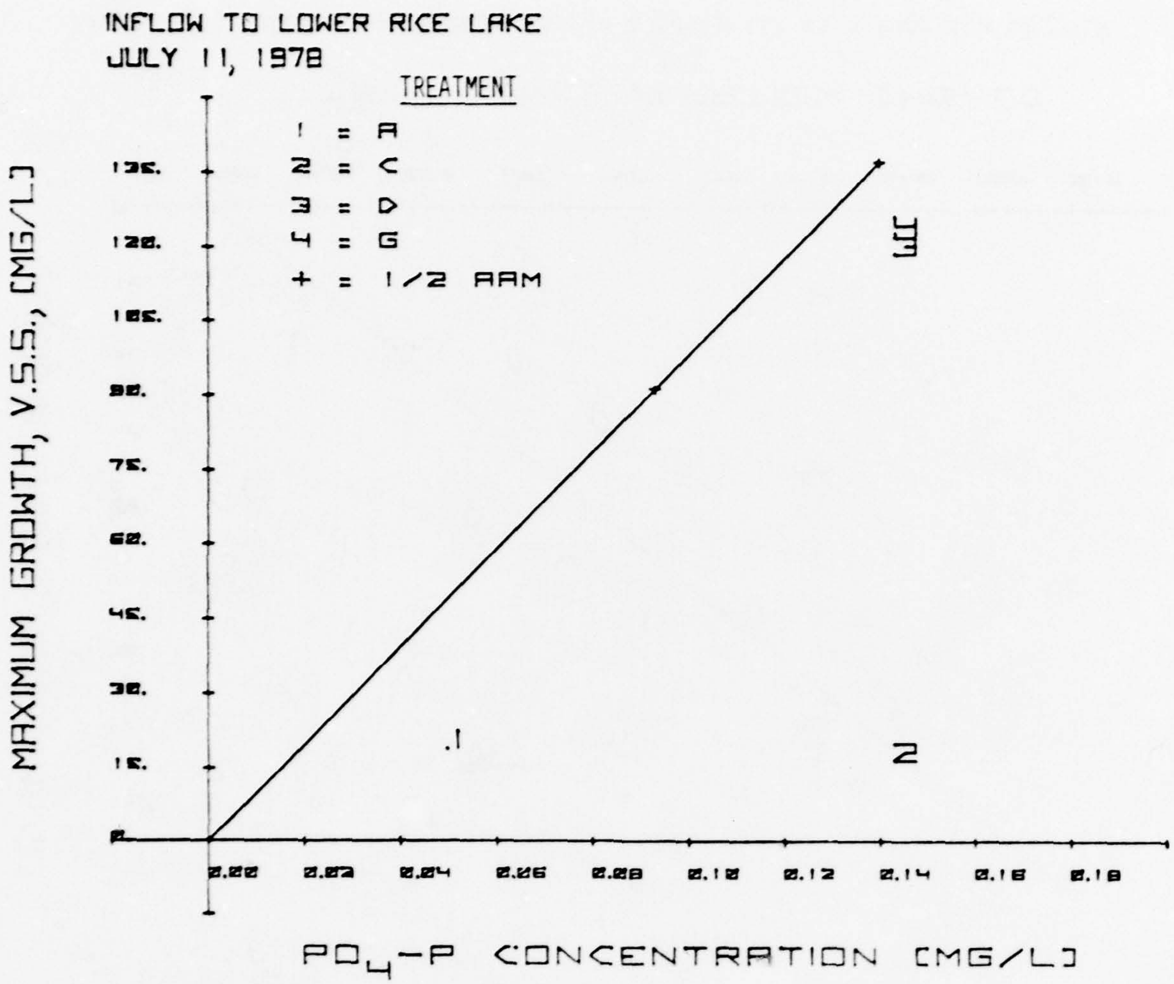


Figure A33. Maximum growth in sample 2 (July 11) as a function of PO_4-P

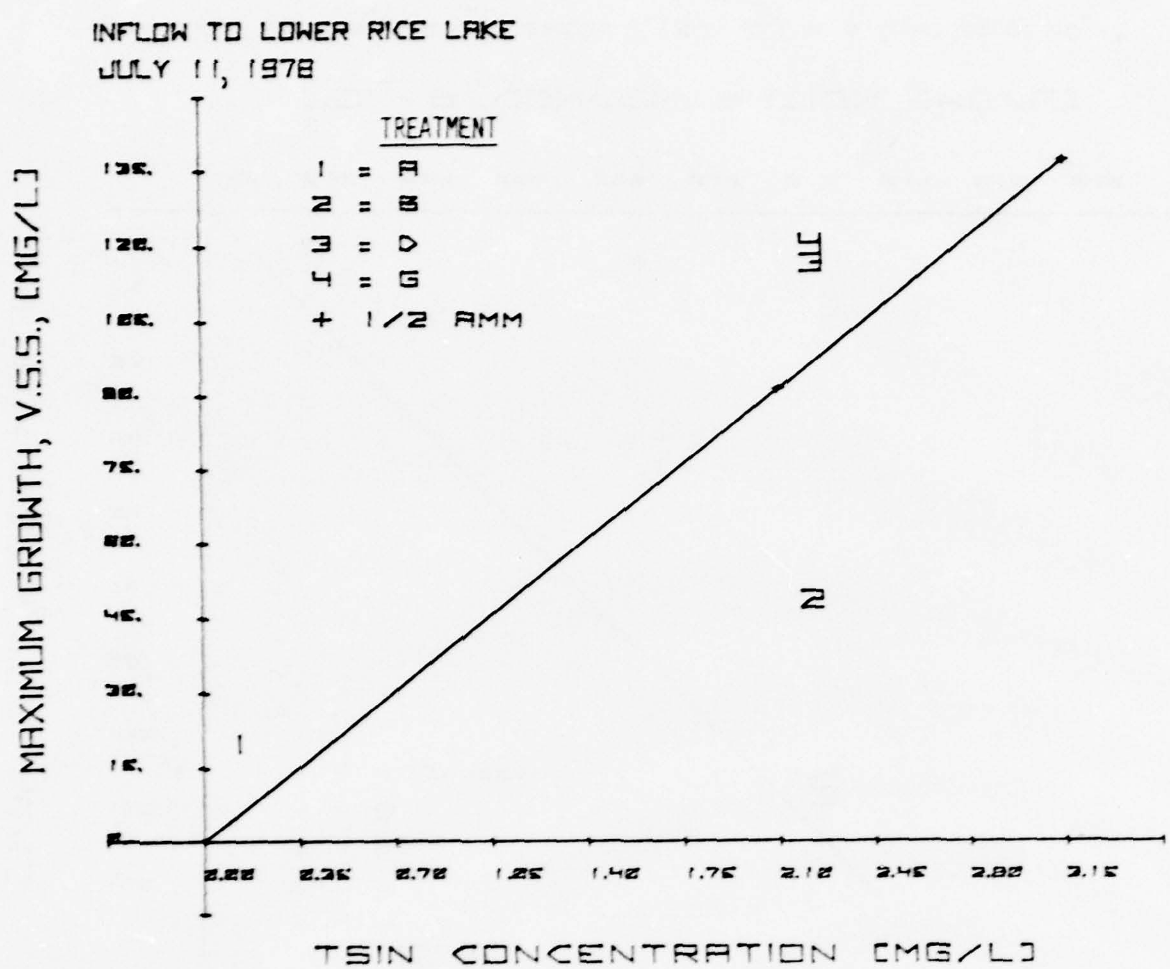


Figure A34. Maximum growth in sample 2 (July 11) as a function of TSIN

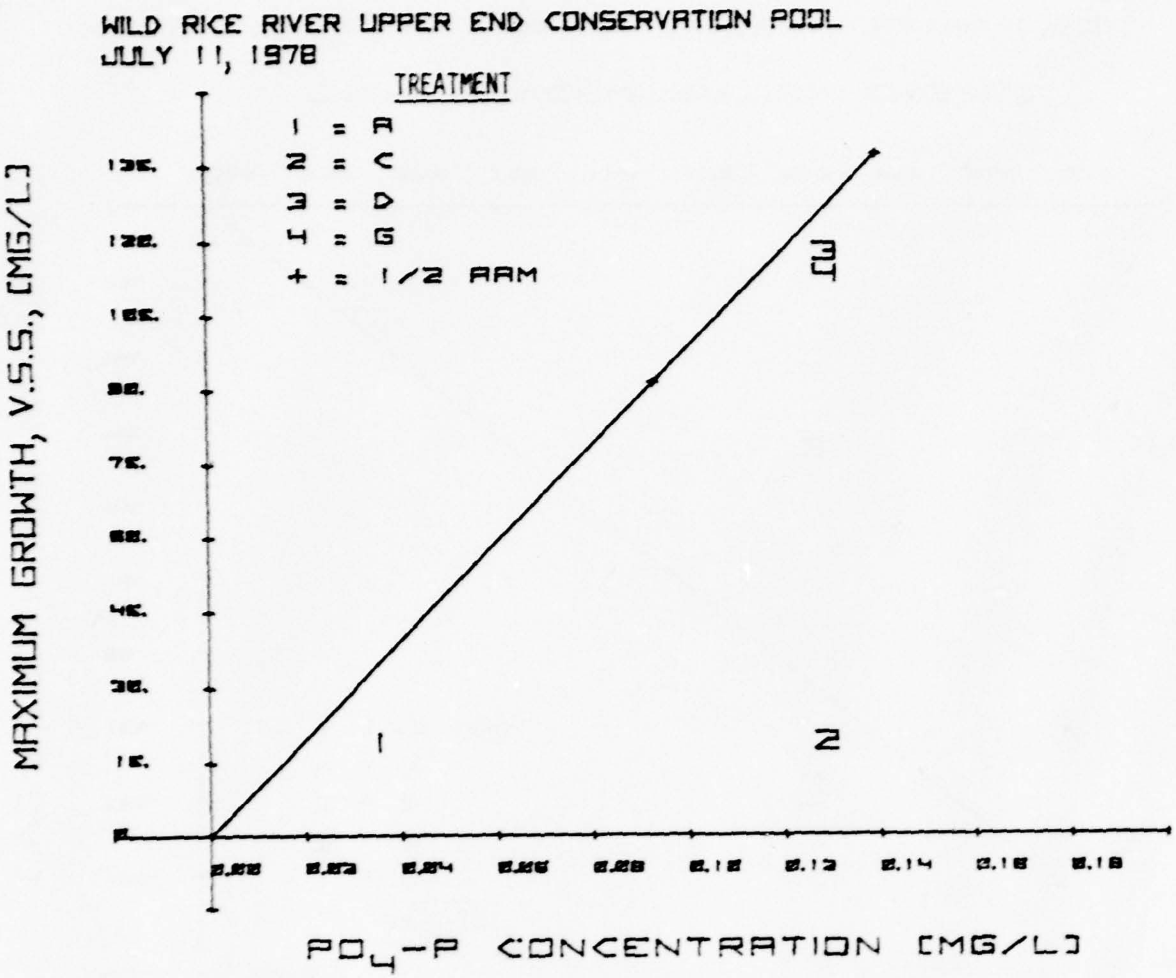


Figure A35. Maximum growth in sample 3 (July 11) as a function of PO_4-P

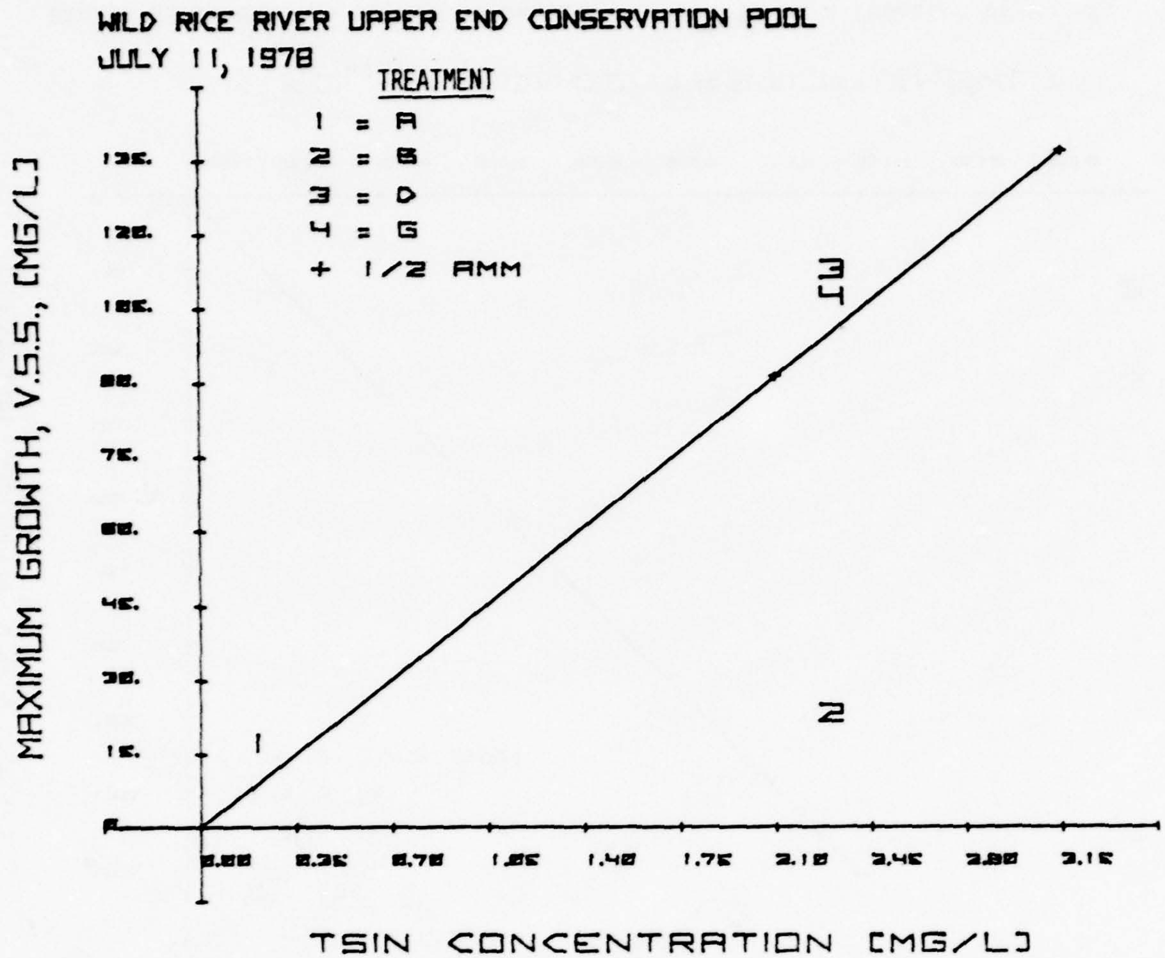


Figure A36. Maximum growth in sample 3 (July 11) as a function of TSIN

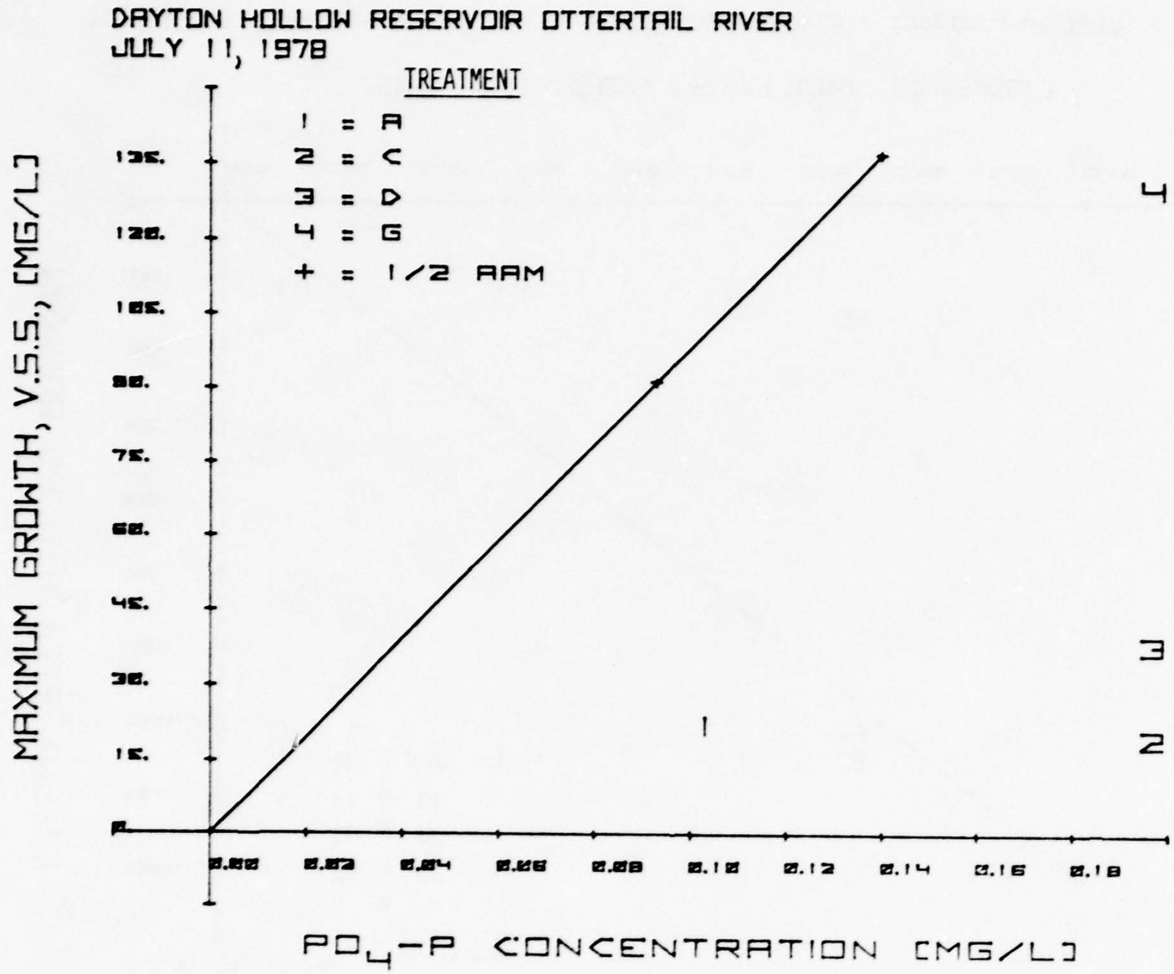


Figure A37. Maximum growth in sample 4 (July 11) as a function of PO_4-P

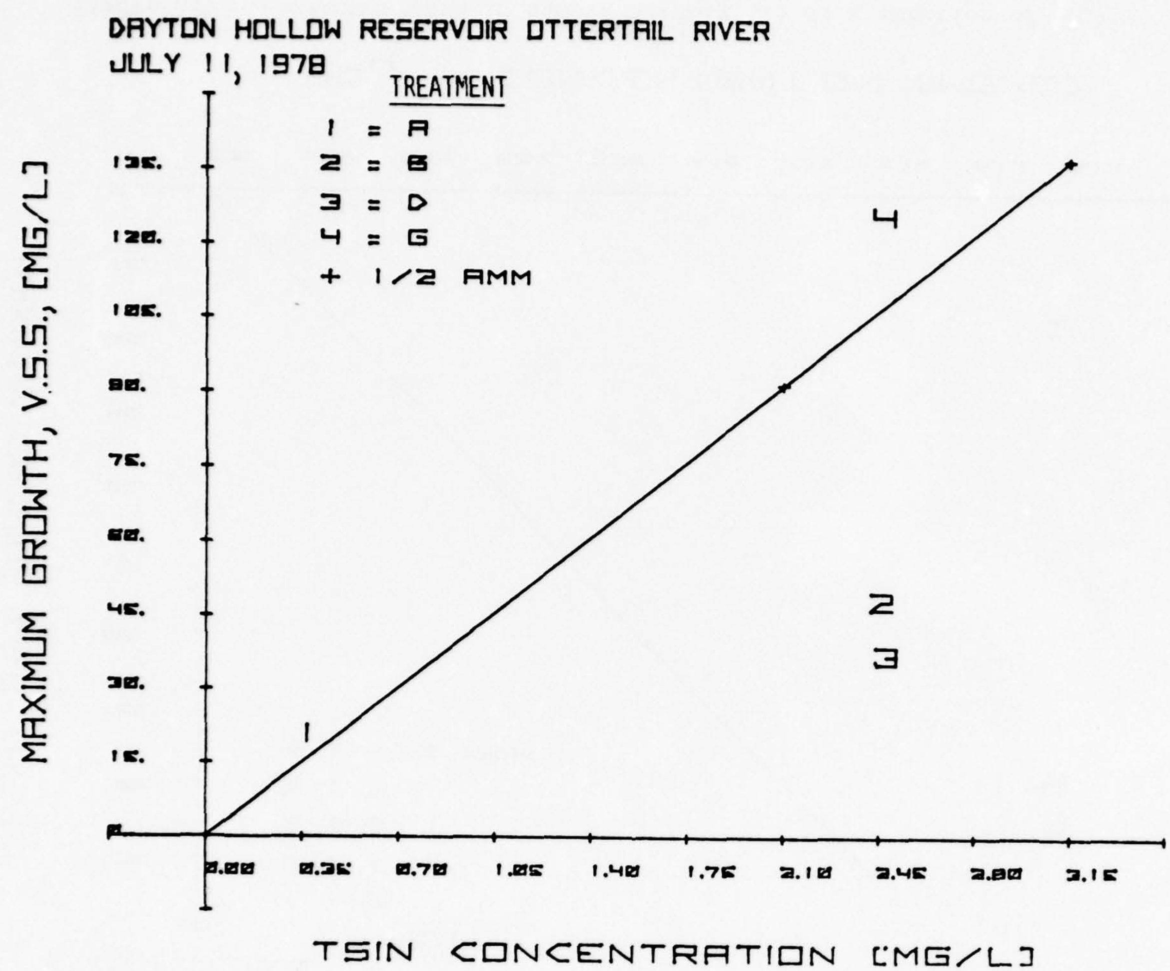


Figure A38. Maximum growth in sample 4 (July 11) as a function of TSIN

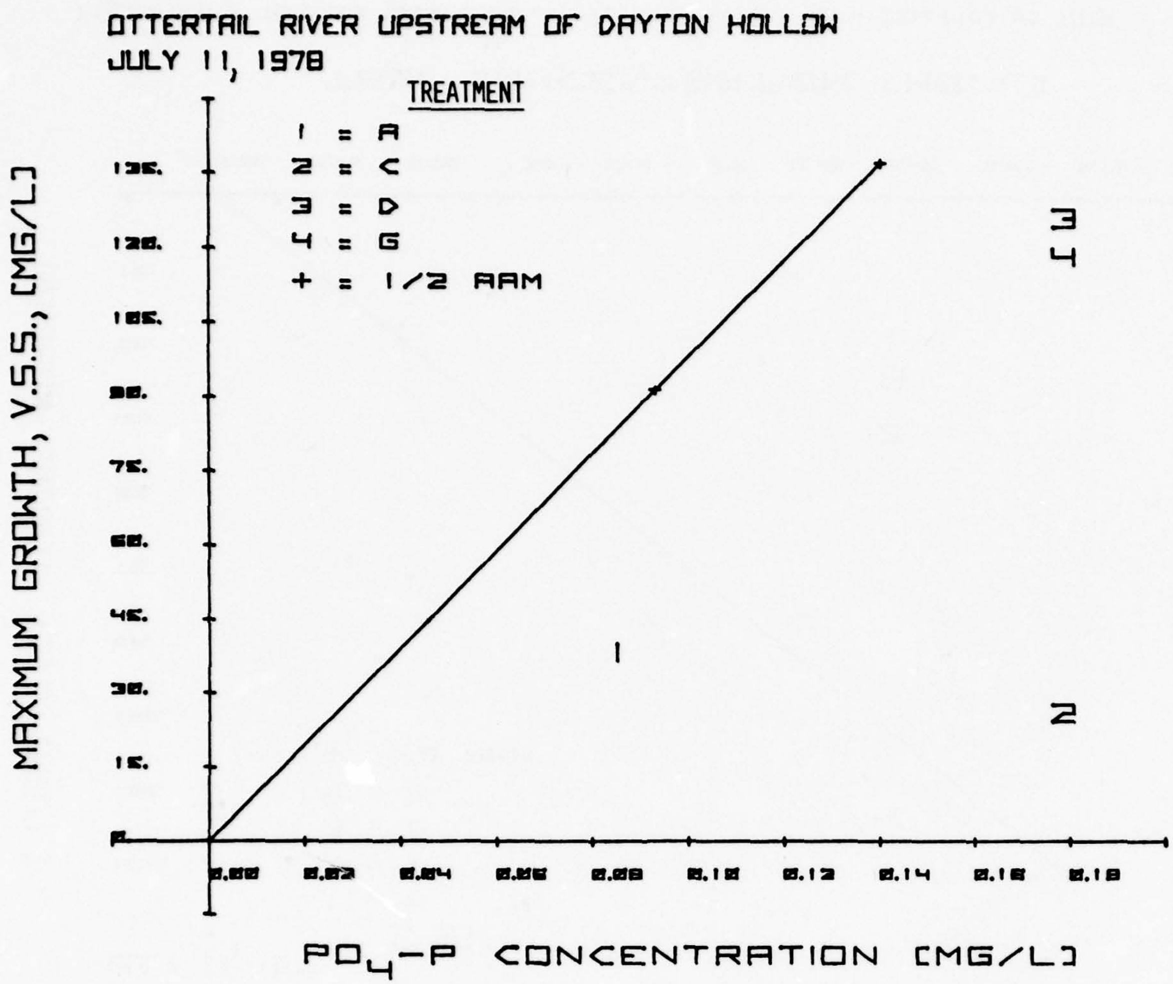


Figure A39. Maximum growth in sample 5 (July 11) as a function of PO_4-P

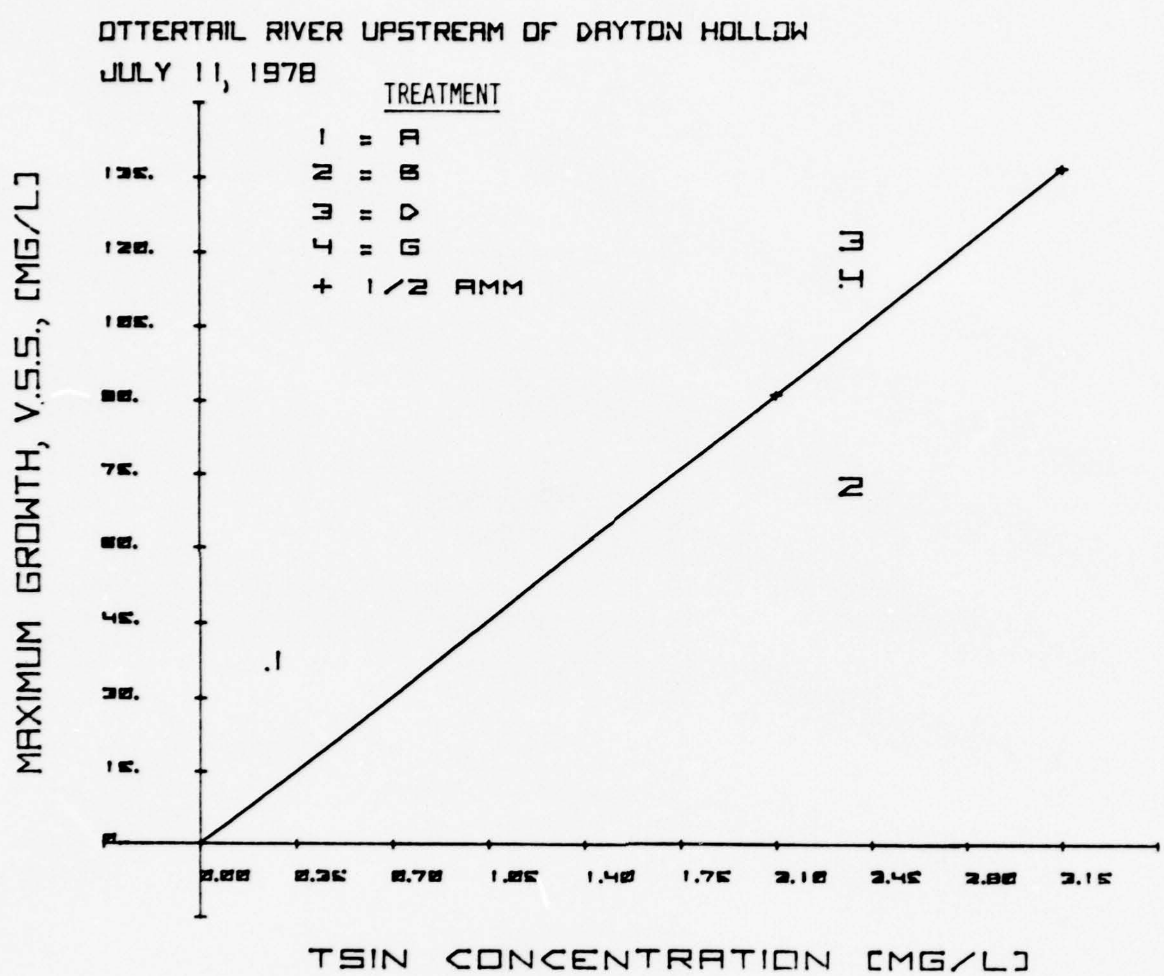


Figure A40. Maximum growth in sample 5 (July 11) as a function of TSIN

APPENDIX B:

A DETERMINATION OF POTENTIAL WATER QUALITY CHANGES IN THE BOTTOM WATERS DURING THE INITIAL IMPOUNDMENT OF THE PROPOSED TWIN VALLEY LAKE

by

D. Gunnison, J. M. Brannon, I. Smith, Jr.,
G. A. Burton, and P. L. Butler

Environmental Laboratory
U. S. Army Engineer Waterways Experiment Station

PART I: INTRODUCTION

1. The objective of this study was to evaluate the potential geochemical effects of soil-water interactions occurring under anoxic conditions on the water quality of the proposed Twin Valley Lake project, Wild Rice River, Minnesota. Consideration was also given to the potential effects of several alternative clearing and filling practices on the water quality characteristics of the project. For the purpose of this report, the water quality characteristics of major concern include: dissolved oxygen (DO) and biochemical oxygen demand (BOD), pH, nutrients of major importance in supporting algal growth, sulfide, organic carbon, color, and the metals iron and manganese.

PART II: METHODS AND MATERIALS

Selection, Characteristics, and Sampling of Soils and Vegetation Used in this Study

2. Two generally representative areas from within the boundaries of the proposed Twin Valley Lake were selected as sampling sites. The general locations of these sites are depicted by Arrows 1 and 2 in Figure B1. Major vegetative and edaphic considerations for each of the sites are presented below.

3. Site 1 was selected as representative of the most extensive plant community and soil type lying within the proposed lake boundaries. The site is approximately 1.6 km north of County Highway 31 and 61 m north-northeast of the northern terminus of County Road 164, Norman County, Minnesota. Vegetation on this site has been characterized as mature floodplain or bottomland forest consisting of a highly developed overstory of tree species and well-developed herbaceous ground cover (U. S. Army Engineer District, St. Paul 1975). The species composition of floodplain forest for this area is given in detail elsewhere (U. S. Army Engineer District, St. Paul 1975). The soil of Site 1 is classified as type Af, alluvial land, frequently flooded and has a capability unit rating VIw-1 - of restricted use and best suited for wildlife habitat (Soil Survey of Norman County, Minnesota 1974). The periodic inundation of the soil in this area is reflected by the water-tolerant nature of the dominant tree species (U. S. Army Engineer District, St. Paul 1975) and by the lack of development of the soil into normal, distinctive soil profiles (Soil Survey of Norman County, Minnesota 1974).

4. Site 2 was selected as typical of the second most abundant soil type within the proposed lake. Site 2 is situated approximately 1.6 km north of County Highway 31 and 1.2 km north-northwest of the junction of County Highway 36 and County Road 183, Norman County, Minnesota. Vegetation on the type of soil found at this site is also of the floodplain type; however, Site 2 had previously been stripped of

vegetation for agricultural purposes. The portion of Site 2 selected for sampling was approximately 10 m from the nearest plowed area and was 2 to 3 years into a secondary succession at the time of sampling. Predominant vegetation included a mixture of grasses plus numerous small shrubs of the genus Ribes (gooseberry) at an estimated density of 15 stems/m^2. The soil of Site 2 is classified as Ad, alluvial land, occasionally flooded, and having a capability unit rating IIIw-5 - soil with high organic matter content and highly fertile, but limited usefulness by poor drainage (Soil Survey of Norman County, Minnesota 1974). The upper or A horizon of this soil was extremely well developed, and the samples commonly indicated a depth of 33 to 38 cm for this horizon.

5. Soil samples were collected by cutting the soil away from the perimeter of an 0.40-m^2 area down to approximately 200 cm beneath the boundary of the A and B horizons. Individual horizons were removed and placed onto individual sheets of 5-mil-thick polyvinylchloride (PVC) plastic prior to placement in shipment containers and transportation to the U. S. Army Engineer Waterways Experiment Station (WES). In the case of Site 1, none of the large trees and shrubs present were included with the soil samples, although any roots running through the various horizons and litter lying on the surfaces of the A horizon samples were included. Samples of twigs and leaves were removed from the vegetation present in the area of Site 1, each in approximate proportion to the biomass of a given species present. Samples were composited and placed into individual heavy-duty 30-gal (0.114-m^3) trash bags for transport to WES. In the case of Site 2, the dominant vegetation present on the site (grasses plus gooseberries) was small enough to permit inclusion with the A horizon samples, and this was done. In addition, samples of the shrubby gooseberries were also taken for independent BOD analysis.

Experimental Setup

6. The general design of the soil-water reaction columns, the instrumentation (system circuit attached) to these columns, and the flow-through system providing constant inflow of synthetic Wild Rice River water are depicted in Figures B2 and B3. Individual samples of soil horizons were trimmed to squares of approximately 0.45 m on each side; these were then each placed into one soil-water reaction column. To prevent surface vegetation and litter from buoying up when the reaction column was flooded, A horizons were covered with one layer of sink matting having a diamond-shaped open mesh of 0.6- by 0.6-cm squares. For each study site, replicates of A horizons were set up in each of three reaction columns.

7. Prior to the initiation of an experiment, the reservoir tanks were filled with deionized water, and each of the compounds listed in Table B1 was added to the reservoir at the concentrations indicated. This combination of ingredients was selected as that which most closely simulates the average yearly composition of the Wild Rice River (refer to USGS data Wild Rice River at Twin Valley, Minnesota Water Year (WY) 77). The reservoir water was actively charged with air for a minimum of 24 hr prior to flooding of the soils. Reaction columns were filled with synthetic Wild Rice River water to the overflow point, and the soil-water contents of each unit were permitted to equilibrate for 1 week with constant aeration and mixing. At this time, an initial sample was taken to provide baseline data under aerobic conditions. After sampling, aeration was discontinued, and the reaction columns were sealed off from the atmosphere. Flow-through conditions were initiated at a rate approximating a 1-year residence time for the water in the reaction column. The ambient incubation temperature was 20°C, and the circulation pump achieved a complete turnover of reaction column water once every 2 min.

8. Reaction columns were run steadily for 120 days and sampled for the various physical and chemical parameters except DO at 0, 1, 2, 5, 7, 9, 13, 16, 21, 30, 40, 50, 60, 75, and 100 days. Dissolved oxygen

was measured daily from the initiation of the experiment up to the point where it was no longer detectable in samples from any of the reaction chambers (19 days). At this point, measurement of oxidation-reduction potential was initiated.

9. After 120 days, water from each of four reaction units was removed and replaced with an equivalent volume of fresh water (approximately 200 ℓ). At this time, one of the columns containing an A horizon from each of the two study sites was completely emptied and replaced with a 15-cm-deep B horizon from Site 2. These two replicates were then flooded with synthetic river water and treated in the same manner as the remaining four A horizons. Also at this time, the inflow-outflow rate was increased to give an average residence time of 35 days, the maximum flow rate that could be achieved with the current flow-through apparatus.

Measurement of DO, pH, Conductivity, and Color

10. Dissolved oxygen, pH, and conductivity were all measured on one 300-ml sample that was collected under nitrogen by permitting water to flow gently from a reaction column sampling port into a standard BOD bottle. Dissolved oxygen and sample temperatures were determined with a YSI Model 57 Dissolved Oxygen Meter equipped with a YSI Model 5720 BOD oxygen probe (Yellow Springs Instruments, Yellow Springs, Ohio). Conductivity was measured with a YSI Model 31 Conductivity Bridge using a YSI Model 3403 Conductivity Cell (Yellow Springs Instruments, Yellow Springs, Ohio). pH was determined with a standard pH meter, while color was analyzed using the spectrophotometric procedure given in Standard Methods (1971).

Measurement of Oxidation-Reduction Potential

11. Oxidation-reduction (redox) potential was measured using the ± millivolt scale of a standard millivolt-pH meter. Electrodes for measurement of redox potential were prepared using the procedures of

Mann and Stolzy (1972). Reference potentials were supplied by a standard pH calomel reference electrode. Each reaction column had one redox electrode reference electrode set mounted in the instrument box located in the circulation pump circuit (Figures B2 and B3).

Sample Collection, Preservation, and Analysis

12. All procedures were conducted under a nitrogen atmosphere to maintain the anaerobic integrity of the samples. Samples to be analyzed for soluble nutrients or for total inorganic carbon (TIC) were cleared of particulate matter by passage through a 0.45-μm membrane filter. Samples for particulate plus dissolved organic carbon were not filtered. Samples for metals analysis were passed through 0.10-μm filters, a treatment shown to remove all particulate and colloidal metals (Kennedy et al. 1974). Samples for total sulfide were taken and preserved simultaneously using zinc; analysis was conducted immediately using the titrametric method given in Standard Methods (1971).

13. Samples for total or soluble nutrients were preserved by immediate freezing and storage at $-40^{o}C$. Samples for TIC analysis were stored at $4^{o}C$ in 10-ml serum vials. Metal samples were preserved by acidification with concentrated (11.6 N) HCl added at the rate of 0.2 ml of acid per 15 ml of sample.

14. Metal concentrations were determined using direct flame aspiration with a Spectrometrics Spectraspan II Ecelle Grating Argon Plasma Emission Spectrophotometer (Spectrametrics, Inc., Andover, Massachusetts).

15. Ammonium-N and orthophosphate concentrations in extracts were determined using a Technicon Autoanalyzer II (Technicon, Inc., Tarreytown, New York).

16. Sulfate concentrations were determined turbidimetrically following conversion of sulfate ion to a barium sulfate suspension.

Soil Characterization

17. Prior to any soil analysis, representative amounts of soil samples from each of the study sites were air dried and then ground to pass a 120-mesh sieve.

Water extract

18. A 40-g subsample of each soil sample was weighed into a 500-ml centrifuge bottle containing 200 ml of deionized-distilled water mixture. The mixture was shaken mechanically for 1 hr and centrifuged at 6000 rpm for 10 min. The resulting supernatant fluid was vacuum filtered through 0.45-μm membrane filters, and this filtrate was immediately frozen at $-60^{\circ}C$ until analyzed for Ca, Mn, Cl, K, and SO_4.

Ammonium acetate extract

19. A 20-g subsample of each soil sample was weighed into a 250-ml centrifuge bottle containing 100 ml of 1 N ammonium acetate (pH 4.8). The mixture was mechanically shaken for 1 hr, centrifuged as described above, and then filtered through 0.45-μm membrane filters. Resulting filtrates were acidified to pH 1 with HCl and stored in polyethylene bottles for subsequent analysis.

Hydroxylamine hydrochloride extraction

20. A subsample of each soil (200 g dry weight) was weighed into a 250-ml centrifuge bottle containing 100 ml of 0.1 M hydroxylamine hydrochloride - 0.01 M nitric acid solution (Chao 1972). The mixture was mechanically shaken for 30 min and centrifuged as previously described. Supernatant fluids were filtered through 0.45-μm membrane filters prior to acidification with HNO_3 to pH 1 and storage in polyethylene bottles for subsequent analysis.

Total digestion

21. A 2.0-g subsample of soil was weighed into a Teflon beaker, and 25 ml of 8 N HNO_3 was added to this. The mixture was heated for 1 hr at approximately $82^{\circ}C$ on a hot plate (Carmody et al. 1973). The extract was then filtered through Whatman No. 5 filters, brought to a final volume of 50 ml with distilled water, and stored until analyzed.

Potassium chloride extract

22. A 20-g soil subsample was weighed into a 250-ml centrifuge bottle containing 100 ml of 1 N KCl. The mixture was mechanically shaken for 1 hr, centrifuged as previously described, and then filtered through 0.45-μm filters. The filtrate was acidified to pH 1 with concentrated HCl and stored in polyethylene bottles until analysis.

Cation exchange analysis

23. A 2.0-g subsample of each soil was saturated with ammonium by shaking for 1 hr with 1 N ammonium acetate. Excess ammonium was removed by repetitive washing with isopropyl alcohol (Jackson 1958). The adsorbed ammonium was then removed by extraction with a series of 2 N solutions of mixed K and Ca nitrates (1.2 N KNO_3 and 0.8 N $Ca(NO_3)_2$, respectively) (Tucker 1974).

Total Kjeldahl nitrogen

24. A 0.5-g subsample of each soil was weighed into a micro-Kjeldahl flask containing 1.1 g of catalyst (100 g of K_2SO_4, 10 g of $CuSO_4 \cdot 5\ H_2O$, and 1.0 g of Se ground together), 2.0 ml of H_2O, and 3.0 ml of concentrated H_2SO_4. The mixture was heated for 5 hr after the digest had cleared. The digest was then allowed to cool, diluted with distilled water, and then filtered quantitatively through Whatman No. 5 filter paper into a 50-ml volumetric flask. This solution was then stored for subsequent NH_4-N analysis.

Carbon

25. Total organic carbon was estimated by weight loss after heating 10 g (oven dry weight) of soil for 8 hr in a muffle furnace at 400°C (Allison 1965). Inorganic carbon content was determined by treating 5.0 g of a soil subsample with 3 N HCl and the measuring the decrease in weight resulting from CO_2 loss (Allison et al. 1965).

Analytical methods

26. Concentrations of iron, manganese, potassium, and calcium were determined using direct flame aspiration with a Spectrametrics Spectraspan II Ecelle Grating Argon Plasma Emission Spectrophotometer (Spectrametrics, Inc., Andover, Massachusetts). Ammonium-nitrogen, orthophosphate phosphorus, and sulfate were assessed according to the

procedures given in the preceding section on sample collection, preservation, and analysis.

Biochemical Oxygen Demand

27. Biochemical oxygen demand (BOD) was determined for triplicate subsamples of each soil sample according to the procedures described in Standard Methods (1971) with the following modifications. To each 300-ml standard BOD bottle was added either (a) a 0.1-g subsample of soil, (b) a 0.10-g subsample of soil along with 5.0 ml of glucose-glutamic acid standard check solution, (c) 5.0 ml of glucose-glutamic acid standard check solution only, or (d) no soil or standard check solution. Following the filling of each bottle with dilution water and the stoppering of each bottle, a standard incubation and DO determination was carried out (Standard Methods 1971). The BOD of individual soil samples was determined by difference, and the results were extrapolated to a milligram of dissolved oxygen consumed by a gram of substrate in a litre of assay water.

28. Oxygen demand of vegetation for each site was measured on 0.10-g subsamples of vegetation from each site. As described previously, samples of vegetation were composited in proportions representative of the site from which they were taken; these were dried to constant weight at 80°C and then ground in a Wiley Mill. To provide an inoculum of decomposer microorganisms, 0.10 g of soil from the site of origin of the vegetation was included in each of several replicates of vegetation. Vegetation BOD was determined by difference between samples containing vegetation plus soil and samples that contained soil only.

PART III: RESULTS

29. Results obtained during the present study are reproducible, although initial difficulties with the control of the flow-through system did result in some large variations. These difficulties were removed by the end of the first simulation, and experimentation following the second flooding of the four remaining A horizons and the first flooding of the two B horizons was accomplished under much more controlled conditions. The results obtained in the first simulation generally reflect the trends encountered during the onset and continuation of anaerobic conditions as observed both in natural ecosystems and in bench-scale laboratory microcosms (for details of the general significance of these trends, see Brannon et al. 1978). Generally, there were no significant differences between the geochemical data obtained from the A horizons of the two study sites, and, except where specifically mentioned, the data presented represent averages of the results obtained with all six reaction columns.

Change From Aerobic to Anaerobic Conditions

Biochemical oxygen demand

30. Values for BOD for the A horizons and for the composited vegetation from the two sites are summarized below:

	Site 1	Site 2	F-Value
BOD of A Horizon (mg $O_2/(\ell \times g)$ substrate)	12.3(0.8)*	9.6(1.1)	0.68
BOD of Composite Vegetation (mg $O_2/(\ell \times g)$ substrate)	42.5(8.2)	28.8(7.9)	1.94

* Values given are averages of a minimum of three repetitions. Values in parentheses are standard error of the mean values in a 5-day test.

The F-values required for significant differences between Site 1 and Site 2 at the 95 percent confidence level are 6.61 for the A horizons and 7.71 for the vegetation, respectively. Thus, there is no significant

difference between the two sites for the BOD's of either the soil or for the vegetation. Therefore, the average BOD of the A horizon is 9.8 (standard error = 1.3) mg/(ℓ × g) per gram while that of the vegetation is 35.2 (standard error = 5.6) mg/(ℓ × g) per gram.

Dissolved oxygen

31. Depletion of DO in the water columns of the reaction chambers is given in Figure B4. Aeration was terminated in the columns on day 1 of the incubation period, and active depletion of DO occurred from this point until day 19 at which time DO was no longer detectable in any of the reactors. The reactors became anoxic on an average of 16.5 days of incubation or 15.5 days after the termination of aeration. During this time an average of 8.2 mg of DO was removed from each litre of reaction column water. Since each column contained approximately 200 ℓ of water, a total of 8.2 mg/ℓ × 200 ℓ or 1.64×10^3 mg of oxygen was consumed in 15.5 days for a depletion rate of 106 mg/day. Since the soil was 0.45 m long on each side, the depletion was exerted by 0.203 m^2 of flooded soil surface for an oxygen demand of 522 mg $O_2/(m^2 \times day)$ at 20^oC. This computation ignores the contribution of the sides of the soil sample to the oxygen demand that would tend to depress the demand value; however, the contribution of oxygen by the inflowing water is also ignored (approximately 4.5 mg DO/day), a factor that would increase the demand value.

Oxidation-reduction potential

32. The average oxidation-reduction (redox) potential for the six reactors for the 120-day incubation period is also presented in Figure B4. Examination of the changes in redox potential with respect to time indicates a rapid initial decline corresponding to the period of time encompassing the final exhaustion of DO and the onset of anoxic conditions. Once the initial decline is complete, redox potential levels off and stabilizes somewhat at approximately -300 mV; this falls within the limits of the majority of the standard error bars from day 25 to day 66. In all succeeding figures, the time interval encompassed by days 14 through 19 is referred to as the "transition" stage; this is bounded on the left by aerobic conditions as indicated by the presence

of DO and positive redox potentials and on the right by anaerobic conditions as evidenced by the lack of DO and the existence of negative redox potentials (Figure B4).

pH and Conductivity

33. Changes in pH and conductivity for the 100-day incubation period are shown in Figure B5. The pH decreased rapidly during the first 10 days of incubation, after which a leveling occurred, and the pH decline became nearly linear with a very gentle slope of less than 0.02 pH units/day. The initial highly alkaline pH at 0 time resulted from the buffering capacity provided by the large amounts of $CaCO_3$ and $MgCO_3$ used in the synthetic Wild Rice River water (Tables B1 and B2) in order to simulate the high levels of Ca, Mg, and HCO_3 within the existing river (see USGS data, Wild Rice River at Twin Valley, Minnesota, WY 77). The decline in pH is probably a consequence of the gradual increase in total inorganic carbon (Figure B6) and organic acids occurring during and after the removal of DO.

34. Conductivity increased throughout the entire incubation period, although the increase was more rapid during the first one half of the incubation period, and the conductivity curve then assumed a more gentle but exponential increase towards a final level of 6.0×100 µmhos/cm.

Carbon

35. Changes in both TIC and total organic carbon (TOC) are given in Figure B6. There was no significant difference at the 95 percent confidence level between any of the values for filtered and nonfiltered TOC on a given sampling date after day 1 of the incubation period. Thus, the values shown in Figure B6 are averages of all filtered and nonfiltered samples, and all TOC values shown in Figure B6 are considered to represent soluble constituents.

36. The gradual increase in both TIC and TOC over the initial

50 days of incubation is similar; however, the TIC values are nearly an order of magnitude greater than their TOC counterparts. The TOC concentrations peaked at 50 days of incubation and then began a gentle decline while the TIC concentration continued to increase with increasing length of incubation.

Nitrogen

37. Four forms of soluble nitrogen were monitored throughout the 100-day incubation period. These include: total Kjeldahl nitrogen (TKN), ammonium-nitrogen (NH_4-N), nitrite-nitrogen (NO_2-N), and nitrate-nitrogen (NO_3-N).

Total Kjeldahl nitrogen

38. TKN is shown in Figure B7. NH_4-N is also plotted in Figure B6 because TKN is a measure of the sum of organic nitrogen plus NH_4-N. While the values for TKN are always greater than NH_4-N values for the same sampling date, some changes in the TKN composition are apparent. Throughout the entire aerobic phase of the incubation (0 to 14 days), the amount of NH_4-N present is negligible while that of soluble TKN is in the 2- to 5-mg/ℓ range. Thus, the TKN present initially is predominantly organic in nature. Once the reactor systems have become anoxic, NH_4-N begins to accumulate in a linear fashion. TKN also increases linearly from day 21 to day 60 and then falls off to the point where the values for TKN on day 75 and day 100 are nearly identical to those of NH_4-N for the same days. The difference between the organic nitrogen and the ammonium nitrogen components of the TKN is summarized by the following:

Incubation, days	Percent Organic Nitrogen	Percent NH_4-N
0	100	0
21	83	17
40	83	17
60	71	29
75	21	79
100	25	75

Inorganic nitrogen

39. The various inorganic components of soluble nitrogen are compared in Figure B8. Nitrate-nitrogen, which is considered as an alternate electron acceptor for oxygen during periods of little or no DO, declines rapidly throughout the period of DO depletion, but nitrate does not undergo total depletion until after the transition period wherein DO is exhausted. Nitrite-nitrogen is a two-way product, being produced as an intermediate during the aerobic oxidation of ammonium to nitrate and, conversely, being an intermediate during the anaerobic reduction of nitrate to ammonium. The occurrence of a burst of nitrite during the aerobic phase of incubation tends to support the first of the two pathways and helps to account for the depletion of DO shown in Figure B4 and the failure to accumulate large concentrations of NH_4-N. NH_4-N accumulation becomes nearly linear during the anaerobic phase of incubation, and the accumulation of this constituent demonstrated no sign of stopping, even at 100 days of incubation.

Phosphorus

40. Figure B9 presents the changes found in both total phosphorus and orthophosphate-phosphorus (Ortho-P) during the 100-day incubation period. Both components showed a general increase that commenced with the start of incubation and continued well into the anaerobic phase of incubation. Each component showed an increase in the rate of P release into the water column once the reaction columns had gone anaerobic. Since the total phosphorus content of these samples includes all soluble orthophosphates, condensed phosphates, and organic phosphates, the difference between total phosphates and orthophosphate should give some representation of the amount of the sample composed of condensed and organic phosphates. This is summarized by the following:

Incubation days	Organic + Condensed Phosphate/Phosphorus	Percent Ortho-P
0	100	0
21	83	17
40	83	17
60	71	29
75	21	79
100	25	75

The trend appears to be for Ortho-P to start out forming none of the total phosphorus values and to increase to the point where it comprises approximately two thirds of the total phosphorus present.

Iron and Manganese

41. Figure B10 depicts the changes in soluble reduced iron and manganese during the 100-day incubation period. Manganese reduction appears to precede that of iron by about 5 days, at which point the accumulation of both of these species appears to occur in parallel. However, by the end of the incubation period, manganese accumulation appears to have ceased while iron accumulation is still increasing in almost linear fashion. The apparent depression in iron accumulation between days 30 and 50 corresponds to the period of maximum sulfide accumulation in the water column, and, in fact, a black ferrous sulfide precipitate began to be visible on day 33 of incubation (Figure B11). It is possible that iron was being actively released to the water column during the period of apparent depression, but the removal of iron sulfide by precipitation prevented any accumulation in the water column and may actually have removed more iron than was released during this period. A similar phenomenon involving manganese carbonate may have been responsible for the eventual leveling off in manganese accumulation. However, the formation of insoluble manganous carbonate was not assessed by any of the procedures.

Sulfide

Sulfate removal

42. The reduction of sulfate was inferred by the observation of a disappearance of dissolved sulfate throughout the course of the anaerobic phase of the incubation (Figure B11). No significant decrease in the level of sulfate occurred during the aerobic phase of incubation, and the most extensive disappearance of sulfate did not become apparent until after day 27 of incubation.

Sulfide production

43. Accumulation of sulfide was not evident until after day 27 of incubation, a period corresponding to the maximum rate of sulfate disappearance (Figure B11). The actual maximum level of sulfide accumulation in the water column is rather low; however, the removal of sulfide from the water column through the formation of insoluble ferrous sulfide undoubtedly accounts for a much larger amount of sulfide (see preceding section on iron).

Soil Characterization

44. Values obtained in the soil analysis for water-extractable constituents, for extractable chemical constituents, and for general physical and chemical properties are presented in Tables B3, B4, and B5, respectively. Four general observations are immediately apparent. First, with the exceptions of ammonium acetate extractable iron and manganese, and hydroxylamine extractable manganese (all on Table B4), there are no significant differences between the soil characteristics of Site 1 and those of Site 2. Second, the values obtained for Site 1 are quite variable as indicated by the wide range in the 95 percent confidence intervals around the mean values for each parameter; this indicates a very heterogeneous sample in support of the heterogeneous nature of Site 1 soils as described in Part II. By contrast, the third observation is the relative lack of variation in the values obtained for Site 2 soils as indicated by the generally narrow ranges of the

95 percent confidence intervals around the mean values for each parameter; this indicates a very homogeneous sample. Thus, the general lack of significant differences in the soils from the two sites is largely the result of the heterogeneity of Site 1 samples wherein the standard of error of the mean of these samples becomes so large as to overlap the more homogeneous Site 2 samples; this in spite of the fact that the means of the samples from these sites are often quite far apart. The fourth observation is that the soils contain high concentrations of chemical constituents subject to biological reduction under anaerobic conditions. These include water soluble sulfate (Table B3) and ammonium acetate and hydroxylamine extractable iron and manganese. Sulfate, iron, and manganese solubilized by these extractants are generally considered reducible (i.e., subject to biological reduction under anaerobic conditions (Brannon et al. 1976)).

Color

45. The results of a 1-week extraction of the soils with water under aerated conditions gave the following values for the color of the water:

Wavelength of maximum absorption: 576 nm

Hue: Yellow

Percent luminosity: 98.6

A comparison of the changes in quality or intensity of color that occur in water during sequential floodings is not presented here because, at the time of this writing, the experiments had entered only the first 2 weeks of the second simulation. Thus, an extensive listing of successive color changes is not available at present. More definitive quantitative information will follow shortly.

PART IV: DISCUSSION

Changes in Water Quality Occurring as a Result of Impoundment in the Wild Rice River

46. Results of this study indicated that if the proposed Twin Valley Lake were constructed as described in the U. S. Army Engineer District, St. Paul, Final Environmental Impact Statement of February 1975, the changes that can be expected in the water quality properties of the Wild Rice River should follow the trends presented in the following discussion. This discussion is based on the assumption that the sites sampled are representative of the area to be covered by the lake.

Depletion of DO

47. The biochemical oxygen demands of the soils and the vegetation taken from the two study sites are high and will likely cause a large depletion in the levels of DO of the overlying waters, even though the proposed impoundment will not exhibit strong thermal stratification. The oxygen depletion rates observed for the first year of inundation of the A horizon + litter layer in the present study do fall close to the range observed in certain other reservoirs (e.g., U. S. Army Engineer District, Portland 1978; also the values for the organic soils examined in soil-water contact columns by Sylvester and Seabloom 1965). However, such comparisons made between reservoirs suffer from the overall site-specific properties of the individual reservoirs. Nonetheless, with oxygen consumption rates of 520 mg $O_2/(m^2 \times day)$ for the first year of impoundment, the bottom waters will tend to become anoxic within a short period if the lake becomes stratified with bottom temperatures in the 18^o to 23^oC region. Actual in-lake oxygen depletion times would depend on depth of the water column between the bottom of the reservoir and the hypolimnetic-metalimnetic interface and the nature and fate of organic loadings entering the hypolimnion from the watershed above the reservoir and/or from the epilimnion.

48. Once the study sites have been flooded for a year, the oxygen demand will diminish somewhat owing to the losses of some of the readily

available organic matter through decomposition, leaching, and/or suspension and washout of particulates. However, the present study makes no pretense of accounting for whatever alteration in oxygen demand may occur because of inflow and deposition of inorganic soil components over the existing A horizons, nor does this report examine the potential sustaining effect or increase in oxygen demand that may occur should additional soils of the existing type be washed in and deposited, thus replenishing the existing materials.

49. Results of the present study indicate that if the existing A horizon of the soil remains unaltered by deposition from the first to second season, the oxygen demand will fall from an estimated 520 to approximately 438 mg $O_2/(m^2 \times day)$, a decline of more than 80 mg $O_2/(m^2 \times day)$. Whether the demand will be reduced by a similar extent from the second to the third years of inundation cannot be assessed at this time. Should the bottom waters remain aerobic during the first year of impoundment, a larger decrease in the oxygen demand would tend to occur as a consequence of a more efficient and complete utilization of organic matter under aerobic conditions relative to anaerobic circumstances (Alexander 1977; Brock 1967; Thimann 1963). Given the rate of decrease in oxygen demand observed between the first and second simulations, a minimum of 5 years of anaerobic/aerobic conditions will be required to decrease the oxygen demand to the 110 to 120 mg $O_2/m^2 \times$ day level observed for the first year of inundation of the B horizon.

Release of carbon, nitrogen, and phosphorus

50. Results of the present study indicate that release of organic forms of carbon, nitrogen, and phosphorus from the soil into the water column occurs extensively, even under fully aerated conditions. A release of organic materials from these soils is not surprising in view of the high levels of organic matter originally present. The total organic carbon content of the Site 1 and 2 A horizons averages 6.8 percent; this translates to a total organic matter content of 11.7 percent using the transformation factor of Wilson and Staker (1932). This concentration is an average of the entire A horizon, exclusive of the topmost litter layer, but including all underground macro-organic matter,

and is higher than the average value for Minnesota soils, although well within the range for these materials (Buckman and Brady 1969).

51. The values for the total dissolved organic and inorganic forms of carbon, nitrogen, and phosphorus presented here are not necessarily the actual concentrations that will be achieved in the real system; such final values will, of necessity, be determined by the movement of nutrients from sediment to the water column and by mixing within the water column itself. Since water columns of reservoirs are, under normal stratified conditions, not as well mixed as the reaction columns used for this study, the final concentrations of the component nutrients could be much less than that found in the present studies. In this case, however, the concentration of nutrients would increase drastically towards the bottom of the water column.

52. The maximum levels of organic carbon reported in this study (approximately 90 mg/ℓ) are sufficient to tie up nearly 250 mg/ℓ of DO, assuming all carbon to be metabolizable to CO_2. Thus, even at more dilute concentrations, a capacity to exert a BOD will be present. The nitrogen and phosphorus values present in organic materials after the release of the latter from the soil do not represent as much of a direct contribution to the pool of plant-growth stimulating nutrients as do their inorganic counterparts. If the proposed impoundment does go anoxic during the first year of filling, the subsequent buildup of inorganic nutrients will, up to a period of 50 to 60 days, show gradual increases in inorganic carbon, phosphate-phosphorus, and ammonium nitrogen. These substances, if released downstream or if released to the surface waters during the next period of mixing, represent a potential source of plant growth nutrients. Moreover, the concentrations of ammonium observed herein are high enough to cause difficulties with biological oxygen demands exerted in downstream areas as a consequence of the biological oxidation of ammonium to nitrate and nitrite.

Release of sulfide

53. The sulfate contents of both the inflowing Wild Rice River and the soils to be inundated are high. If the proposed impoundment follows the trends observed in the studies, it may become anoxic and if it

remains anoxic for a number of weeks, there is a strong possiblity that hydrogen sulfide will be released. While the resultant levels of sulfide in the water can be limited to a certain extent by the formation and precipitation of insoluble ferrous sulfide, the possibility cannot be excluded that some of the sulfide will escape with the result that its rotten egg odor may be released from the lake. More likely, however, is the potential release of sulfide (to dissolved and suspended particulate) with any bottom withdrawals made from the reservoir and subsequent odor and oxygen demand problems downstream from the impoundment.

Release of iron and manganese

54. The levels of iron and manganese released into the water column by virtue of the solubility of their reduced forms are not as high as those achieved under anaerobic conditions in other situations (Brannon et al. 1978). Moreover, this study indicates that the reddish coloration which can result from the oxidation of iron when anaerobic waters containing the ferrous ion are released via bottom withdrawals would likely be noticed more from the turbidities created by flowing iron oxyhydroxides than by the actual color properties of the material. The color imparted by the movement of humic materials from soil into the water will likely be more intense than that of iron oxides. Insoluble ferrous sulfides do give a black color, but these tend to precipitate rapidly and would tend to remain in the lake. Once released, such materials oxidize rapidly and the problems resulting therefrom are odors (sulfide) and oxygen demand (BOD and IOD). Color would hardly be as serious a problem as the oxygen demand.

Color

55. The findings of this study indicate that the yellow color acquired by waters that contact soils having high levels of organic matter in the area of this impoundment will be apparent for the first years both in the waters in the impoundment and in releases made from it. This will be true whether or not the waters become anoxic. Results of the present study indicate that color is little influenced by anoxia. The quality of these waters would be considered as poor by industrial

effluent standards if based on color alone (Standard Methods 1971). However, the color should be little or no worse than any of the natural lakes in the same region of the country and would be considered to have only a minor impact on water quality, unless the water is intended to serve as a source of potable water supply; in this case, increased treatment costs would be incurred.

pH and conductivity

56. The pH will decrease under anaerobic conditions, but the huge buffering capacity of the carbonate-bicarbonate buffer system prevents the pH from dropping to unacceptable levels. The increase in conductivity observed in this study indicates a gradual increase in dissolved substances under anaerobic conditions and this is confirmed by the observed increase in inorganic forms of carbon, nitrogen, and phosphorus.

Influence of Site Preparation on Water Quality

Influence of clearing on water quality

57. The results obtained in this study indicate that the BOD of the vegetation from the two study sites is quite high, an average 5-day demand being approximately 35.6 mg O_2/ℓ of water per gram of vegetation. The samples of vegetation were predried and ground in order to obtain a uniformity of substrate to enable site-to-site and vegetation-to-soil comparisons; this would tend to increase the BOD values to some extent because of the effect of increased surface area upon microbial availability and colonization (Alexander 1977; Sylvester and Seabloom 1965). However, the data do fall within the range of values obtained by other investigators (Sylvester and Seabloom 1965; Feng and Hyde 1967). The values for the vegetation are three to four times that exerted by the A horizon samples tested. The common practice of removing vegetation only in the flood pool region where residues of dead trees and shrubs can have negative aesthetic impacts relative to recreation will probably also be desirable for the present impoundment. More specifically, however, the vegetation on the sites examined has a large shrubby and herbaceous component (U. S. Army Engineer District, St. Paul 1975);

thus, the BOD of this material is exerted by substances that are relatively easily decomposed when compared with a mature, climax forest. Removal of bottomland vegetation would considerably reduce the BOD of the sites studied (per square metre basis). This procedure would reduce the project's impacts on water quality, particularly in the first 1 to 3 years after filling.

Influence of soil removal on water quality

58. The A horizons of the study sites together with the litter layers have a large BOD, and this is reflected in the rapid oxygen depletion rates observed in the soil-water reaction units. Removal of the A horizon would decrease the oxygen demand approximately fourfold for the first year of flooding, and although the oxygen demand of the B horizon is still quite high, the lower demand of the B horizon in conjunction with the predicted tendency of the reservoir to undergo intermittent mixing would probably preclude the development of prolonged anoxic conditions. Moreover, preliminary results obtained in the studies of the B horizon suggest that this layer will release a much lower level of plant growth-supporting nutrients to the overlying water column. Note that no attempt is made here to anticipate the amount or nature of A horizon materials that will enter the reservoir from upstream areas and settle in the reservoir. Obviously, materials of a highly organic nature will tend to aggravate the DO depletion; those of a more mineral nature will tend to seal off the bottom of the reservoir after deposition and, thus, would lower any oxygen demand. It should also be noted, however, that the A horizon is relatively deep (25 to 46 cm), rendering removal an extremely expensive proposition.

Influence of filling practices on water quality

59. Because both the color and oxygen demand problems improve upon reflooding and reexposure of the soil to fresh waters, the practice of filling and flushing the impoundment two to three times prior to final filling should have a positive effect on reservoir water quality. However, since much of the aging process depends as much on the breakdown of moderately degradable components (cellulose, hemicellulose) as on the

movement of readily soluble components out of the reservoir, those filling practices which tend to accelerate degradation of organic matter while avoiding severe BOD problems are advisable. This suggests a sequence involving two or three flushings to remove easily soluble or leachable components, followed by slow incremental filling to keep the reservoir shallow for as long as possible to promote oxygen exchange with the atmosphere and consequent efficient decomposition of organic matter.

PART V: CONCLUSIONS AND RECOMMENDATIONS

60. Impacts of reservoir construction and operations will be minimal and limited to additions of color (yellow) and soluble organic forms of carbon, nitrogen, and phosphorus to the water, provided the proposed reservoir does not stratify for more than 8 to 10 days. Under aerobic conditions, the impact of the organic material on water quality is larger in terms of the BOD exerted by the material itself than the concentrations of plant nutrients that may be accumulated. The magnitude of the oxygen demand depends on the residence time of the water in the reservoir. A period of successive initial fillings and flushings should minimize this demand because short residence times should promote good dilution and because soluble and leachable components will be readily removed.

61. Anoxic conditions can potentially develop during the first year if stratification persists for longer than 14 to 19 days. Once anaerobic conditions have developed, both organic and inorganic forms of carbon, nitrogen, and phosphorus predominate for the first few weeks of impoundment, but ammonium nitrogen and ortho-P make up increasingly larger portions of total nitrogen and phosphorus as anaerobic incubation continues. Accumulation of inorganic carbon, ammonium-nitrogen, and ortho-P will continue to increase throughout the entire period of anaerobic incubation. Bottom withdrawals from a reservoir under these conditions would release significant concentrations of inorganic forms of carbon, nitrogen, and phosphorus. These nutrients could also be released to the surface waters during periods of wind induced mixing.

62. After 27 days of incubation (average 11 days of anoxic conditions) sulfide accumulation was observed in the studies. The detection of sulfide in the water column corresponded to the observation of a black precipitate of ferrous sulfide and the rotten egg odor characteristic of hydrogen sulfide apparent in samples taken at this time. Bottom withdrawals under these conditions would have sulfide problems.

63. Studies of the BOD of composite vegetation from both sites studied indicate that the vegetation will have a BOD approximately

four times that of the A horizon of the soil. The shrubby and herbaceous nature of much of the vegetation at the study sites indicates that much of the growth is of an easily decomposable nature. Removal of this bottomland vegetation should improve the oxygen demand, although quantitative data are not available at this time. The initial oxygen demand of samples of A horizon + litter from the study site was approximately 520 mg $O_2/(m^2 \times day)$. This demand decreased approximately 80 mg $O_2/(m^2 \times day)$ after more than 100 days of flooding followed by reaeration for 1 week and exposure to a column of fresh water under a 35-day retention time simulation. By contrast, the oxygen demand of the B horizon from the second study site was less than 120 mg $O_2/(m^2 \times day)$ suggesting that removal of the A horizon could yield a fourfold improvement in the oxygen demand.

64. A series of fillings and flushing prior to the initial filling of the reservoir followed by a period of incremental filling is suggested as an approach to minimize the impact of flooding an area that contains large levels of organic matter in the A horizon of its mineral soil.

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Table B1

Compounds Added to Flow-Through Water Used for Twin Valley Study

Component	Concentration Used mg/ℓ	Cation Concentration mg/ℓ	Anion Concentration mg/ℓ
$CaCO_3$	212.00	84.90	127.00
$MgCO_3$	125.00	36.00	89.00
Na_2SO_4	57.30	18.50	38.70
KC_1	7.55	4.00	3.55
Totals	402	143	258

Table B2

Total Contributions of the Various Constituents of the Flow-Through Water Based on Solubility-Product and Equilibrium Considerations

Ca^{++} = 16.0 mg/ℓ	Cl^- = 3.55 mg/ℓ
Mg^{++} = 36 mg/ℓ	SO_4 = 38.7 mg/ℓ
Na^+ = 18.5 mg/ℓ	CO_3 = 0.75 mg/ℓ*
K = 4.00 mg/ℓ	

* Even though total $CO_3^=$ added was 127 + 89 or 216 mg/ℓ, the solubility was limited due to the high initial pH (8.5-9.5) and the equilibrium $CO_2 \rightleftharpoons HCO_3^- \rightleftharpoons CO_3^=$.

Table B3

Concentrations of Water-Extractable Chemical Constituents in Soils From the Impoundment Area of the Proposed Twin Valley Lake, Minnesota

Constituent	Concentration, μg/g soil* Site 1	Site 2	Calculated F-Value**, †
Calcium	114.0 ± 41.6	72.4 ± 13.2	5.12
Magnesium	22.9 ± 10.3	19.9 ± 5.72	0.37
Potassium	28.1 ± 10.5	16.8 ± 9.5	3.66
Sulfate	95.7 ± 51.8	87.3 ± 25.1	0.12

* Values are expressed in terms of 95 percent level of confidence.
** F-ratio as obtained by sums of squares analysis of variance.
† Required F-value for significant difference at 95 percent level of confidence is 6.61.

Table B4

Concentrations of Extractable Chemical Constituents in Soils From the Impoundment Area of the Proposed Twin Valley Lake, Minnesota

Constituent	Concentration, μg/g soil* Site 1	Concentration, μg/g soil* Site 2	Calculated F-Value**, †
Ammonium acetate extractable			
Iron	98.6 ± 39.1	34.1 ± 3.42	15.5
Manganese	24.6 ± 10.6	7.3 ± 3.6	13.9
Orthophosphate-P	7.8 ± 8.4	9.7 ± 1.8	0.28
KCl extractable			
Ammonium-N	6.4 ± 4.5	4.4 ± 4.13	0.60
Hydroxylamine extractable			
Iron	35.3 ± 38.6	52.8 ± 7.1	1.14
Manganese	271.0 ± 32.8	340.0 ± 49.9	7.64

* Values are expressed in terms of 95 percent level of confidence.
** F-ratio as obtained by sums of squares analysis of variance.
† Required F-value for significant difference at 95 percent level of confidence is 6.61.

Table B5

General Physical and Chemical Properties of Soils From the Impoundment Area of the Proposed Twin Valley Lake, Minnesota

Constituent	Concentration* Site 1	Concentration* Site 2	Calculated F-Value**, †
TKN, μg/g soil	3566 ± 3215	3290 ± 329	0.04
Total iron, μg/g soil	9458 ± 2595	9463 ± 630	0
Total manganese, μg/g soil	393 ± 99.0	411 ± 20.8	0.18
Total phosphorus, μg/g soil	501 ± 117	511 ± 37.4	0.04
Cation exchange capacity meq/100 g soil	22.4 ± 17.7	22.2 ± 1.4	0
TOC, percent	7.1 ± 7.7	6.5 ± 0.2	0.04
TIC, percent	0.7 ± 0.1	0.7 ± 0.1	3.38

* Values are expressed in terms of 95 percent level of confidence.
** F-ratio as obtained by sums of squares analysis of variance.
† Required F-value for significant difference at 95 percent level of confidence is 6.61.

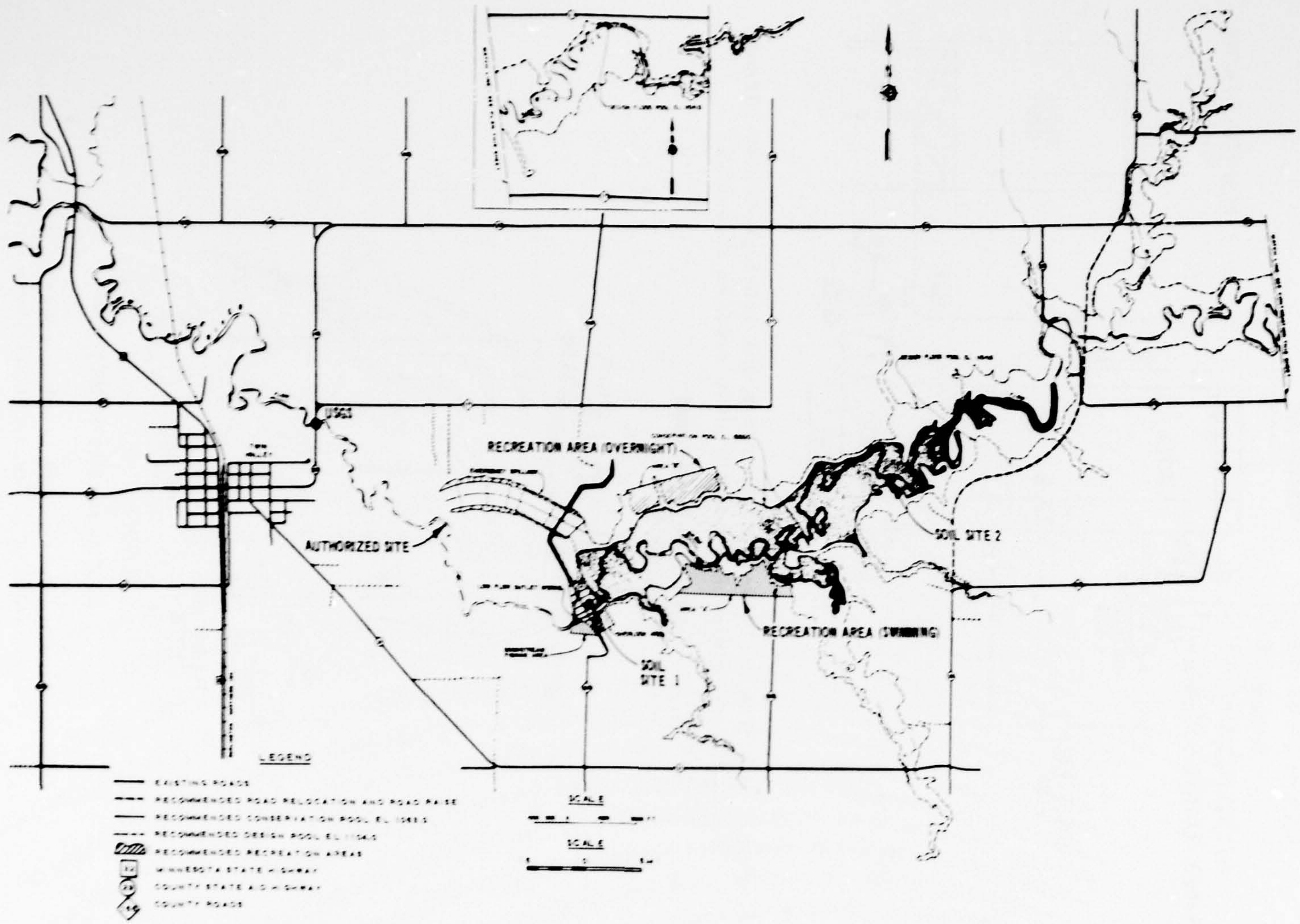

Figure B1. Proposed Twin Valley Reservoir (alternative site)

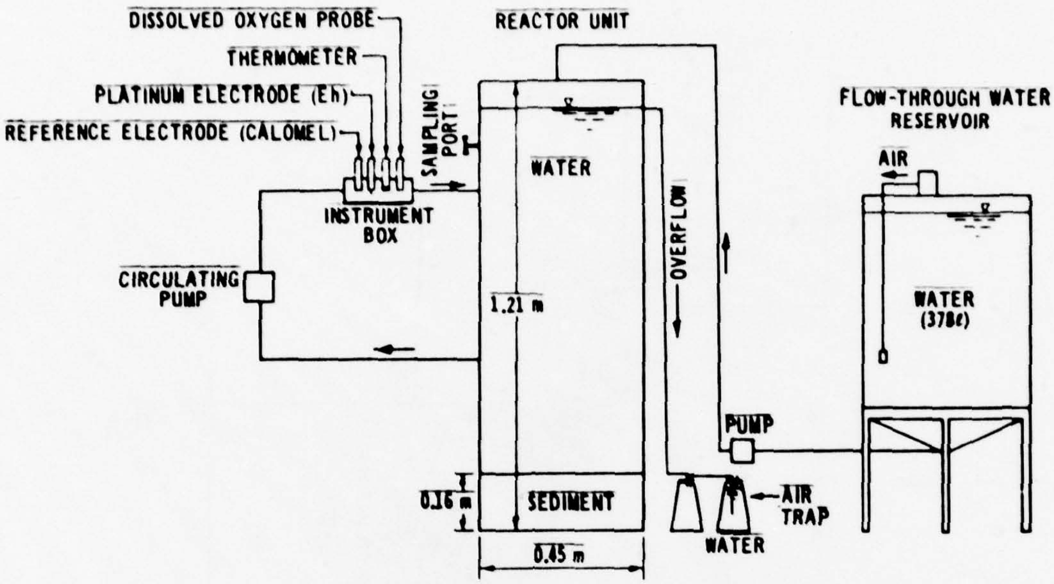

Figure B2. Diagram of flow-through system and instrumentation circuit attached to soil-water reactor unit used in this study

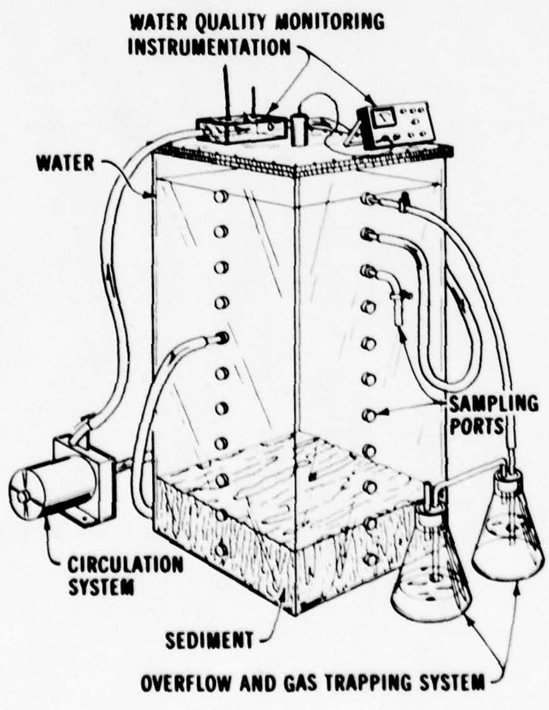

Figure B3. Sediment-water reactor for studying anaerobic conditions in reservoirs

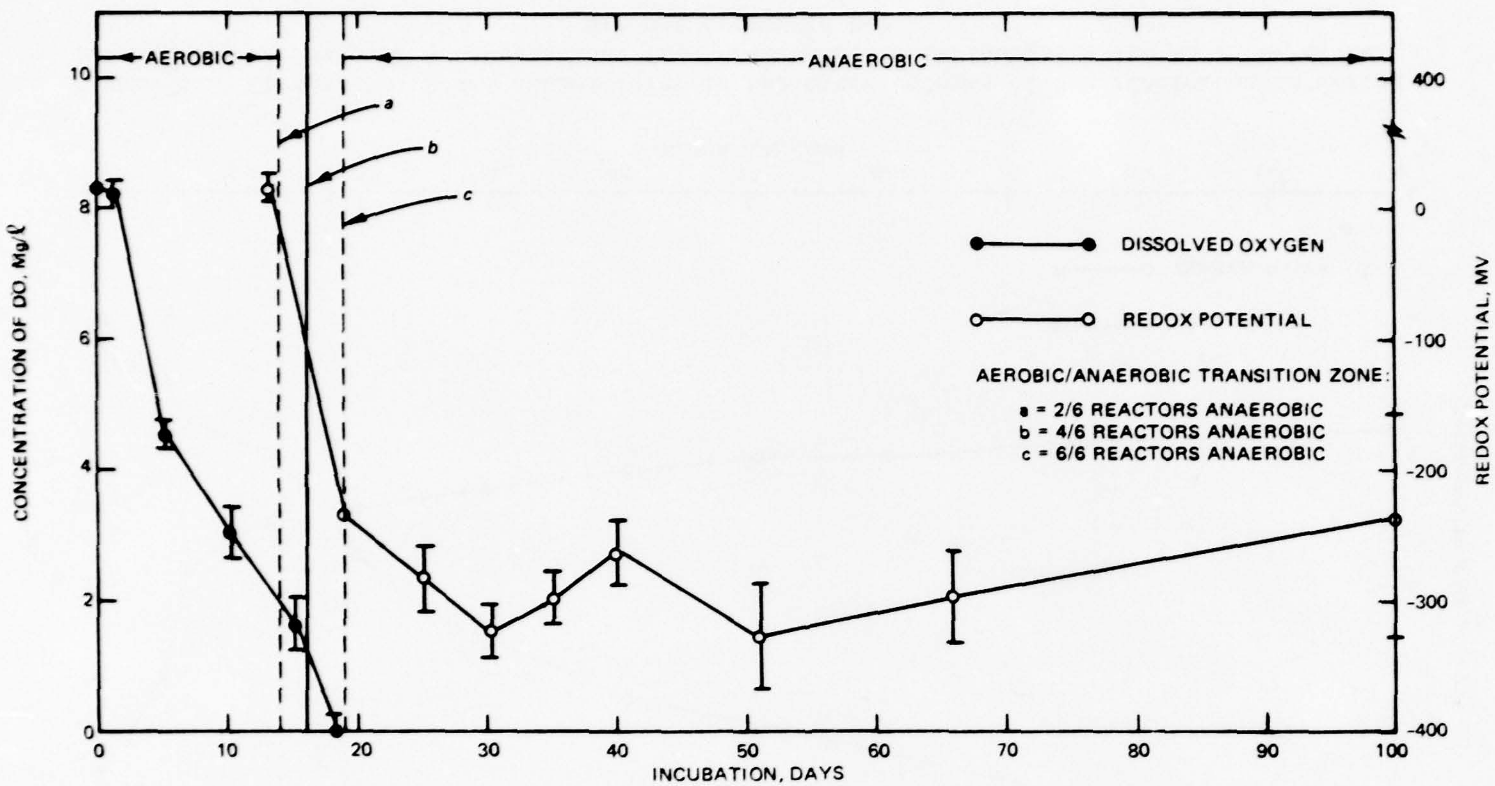

Figure B4. Changes in DO concentration and oxidation-reduction (redox) potential in water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

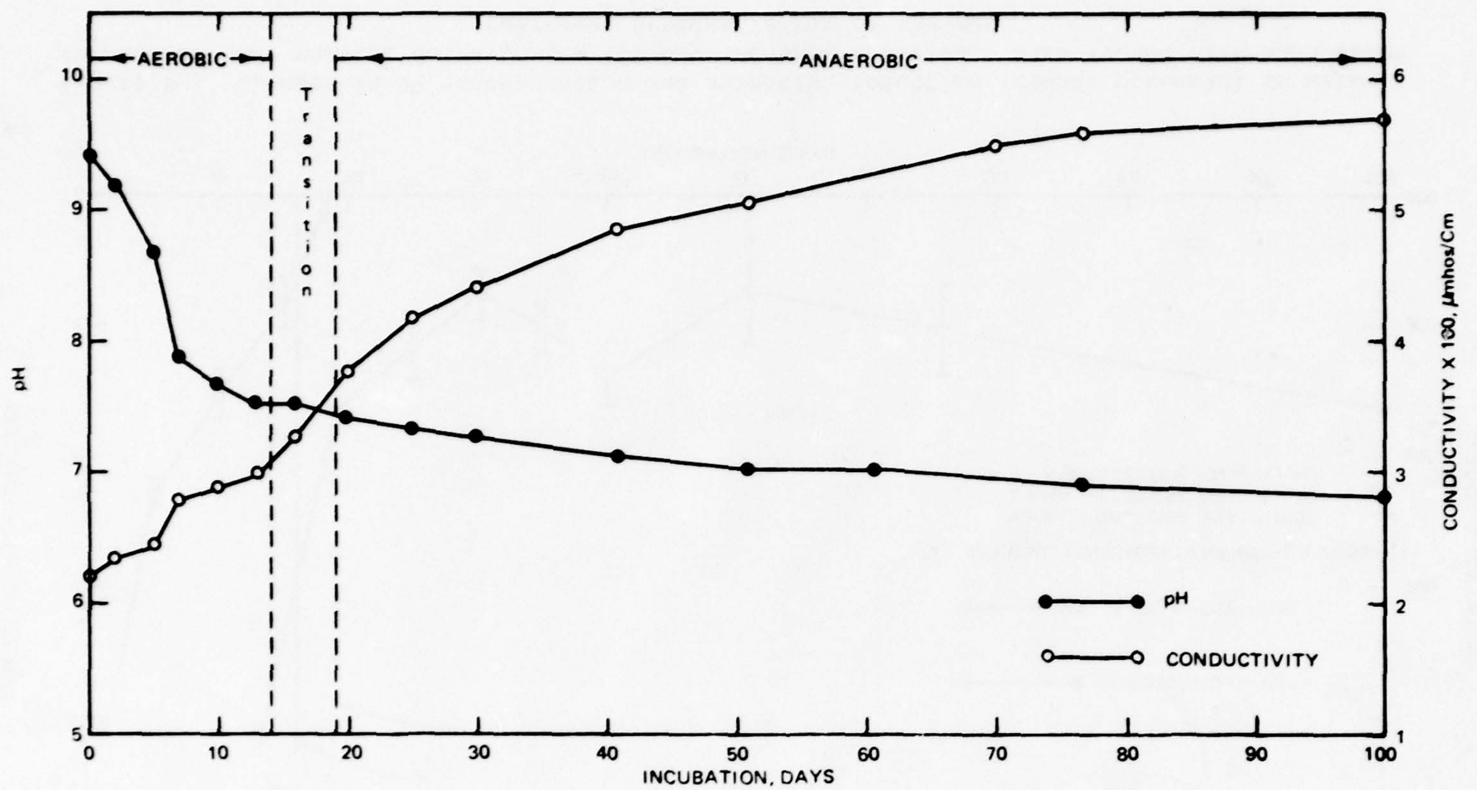

Figure B5. Changes in pH and conductivity in the water columns of the reactor units during 100-day incubation period. Deviations from mean values were insignificant and, therefore, are not presented here

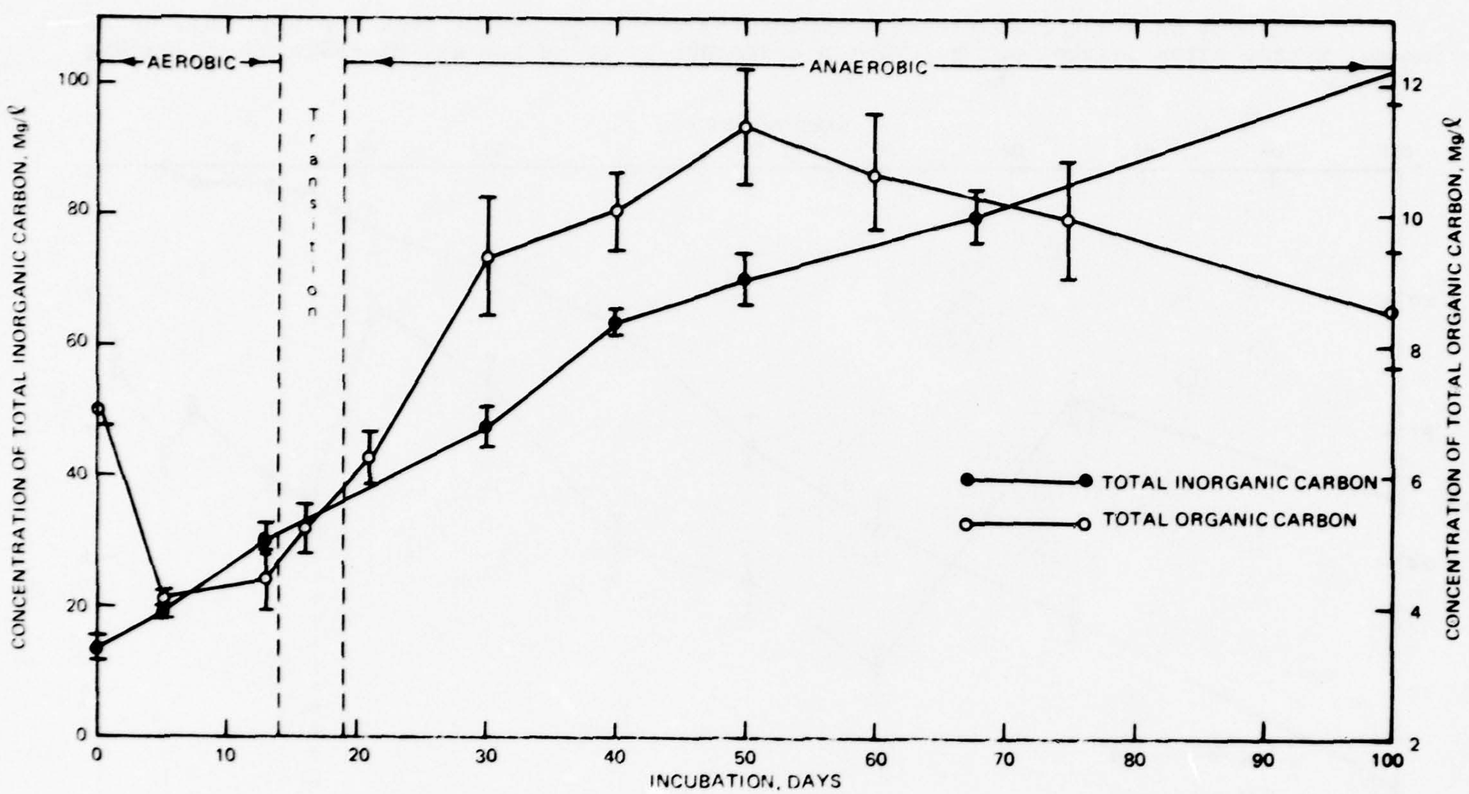

Figure B6. Changes in TIC and TOC in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

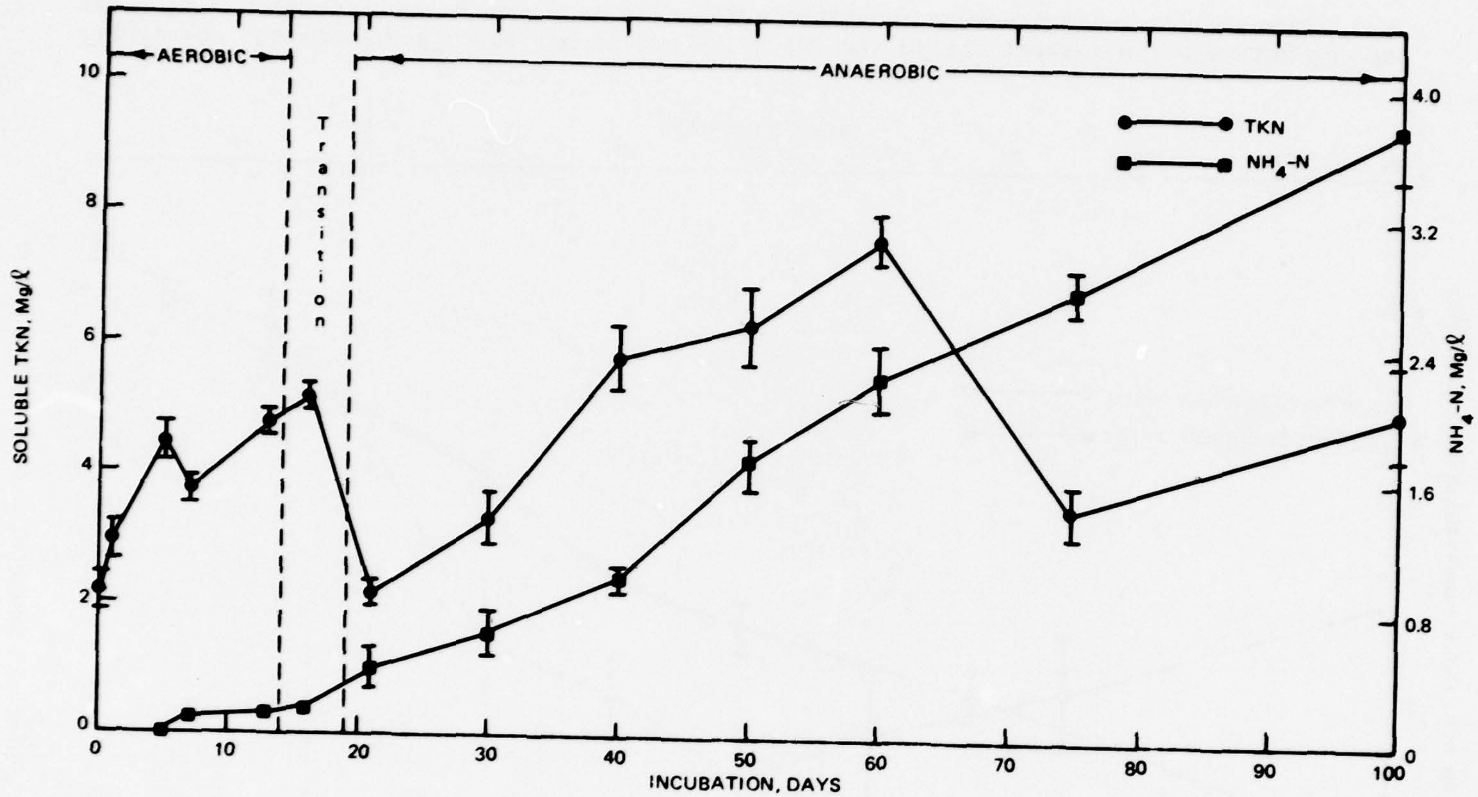

Figure B7. Changes in TKN and NH_4-N in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

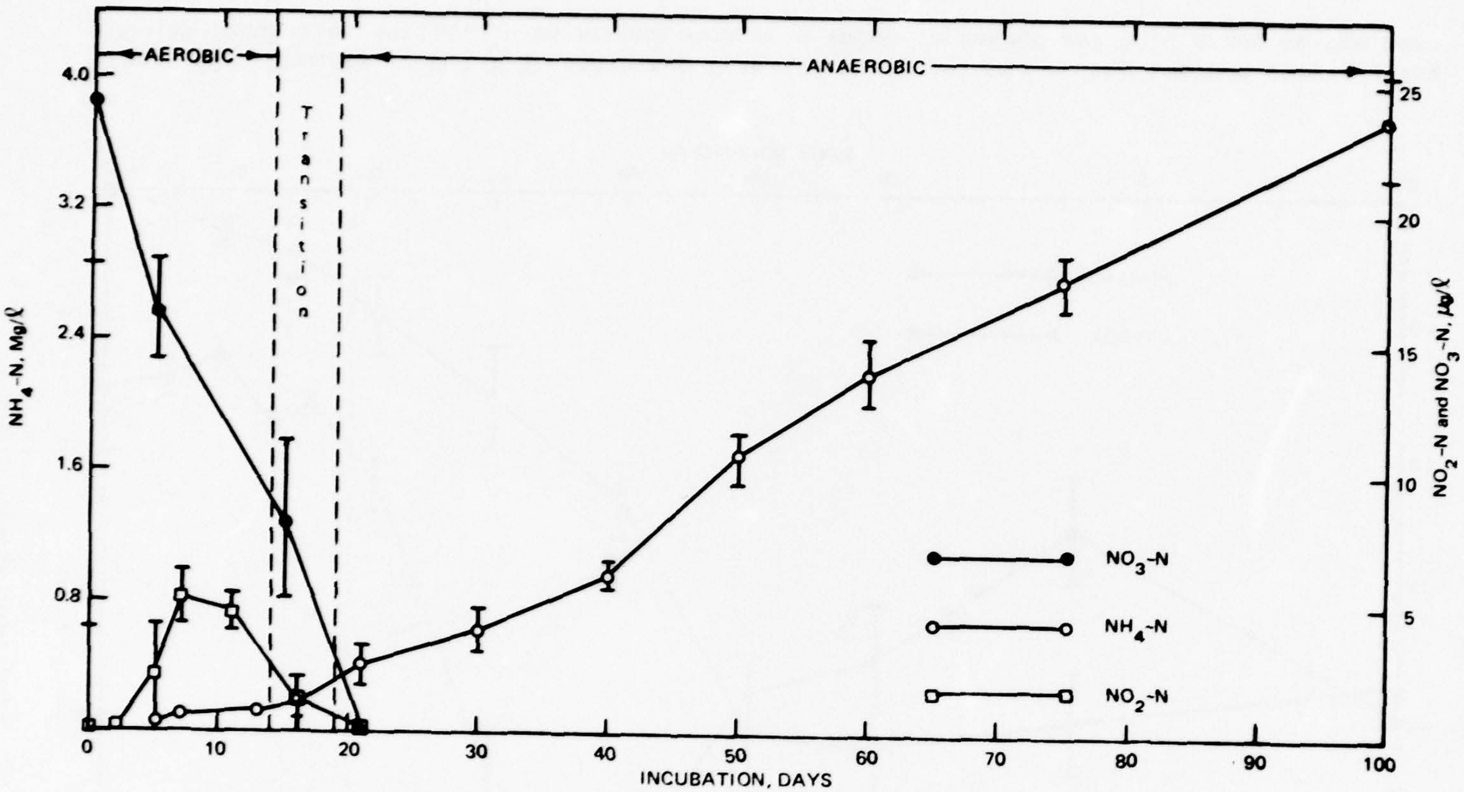

Figure B8. Changes in NH_4-N, NO_3-N, and NO_2-N in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

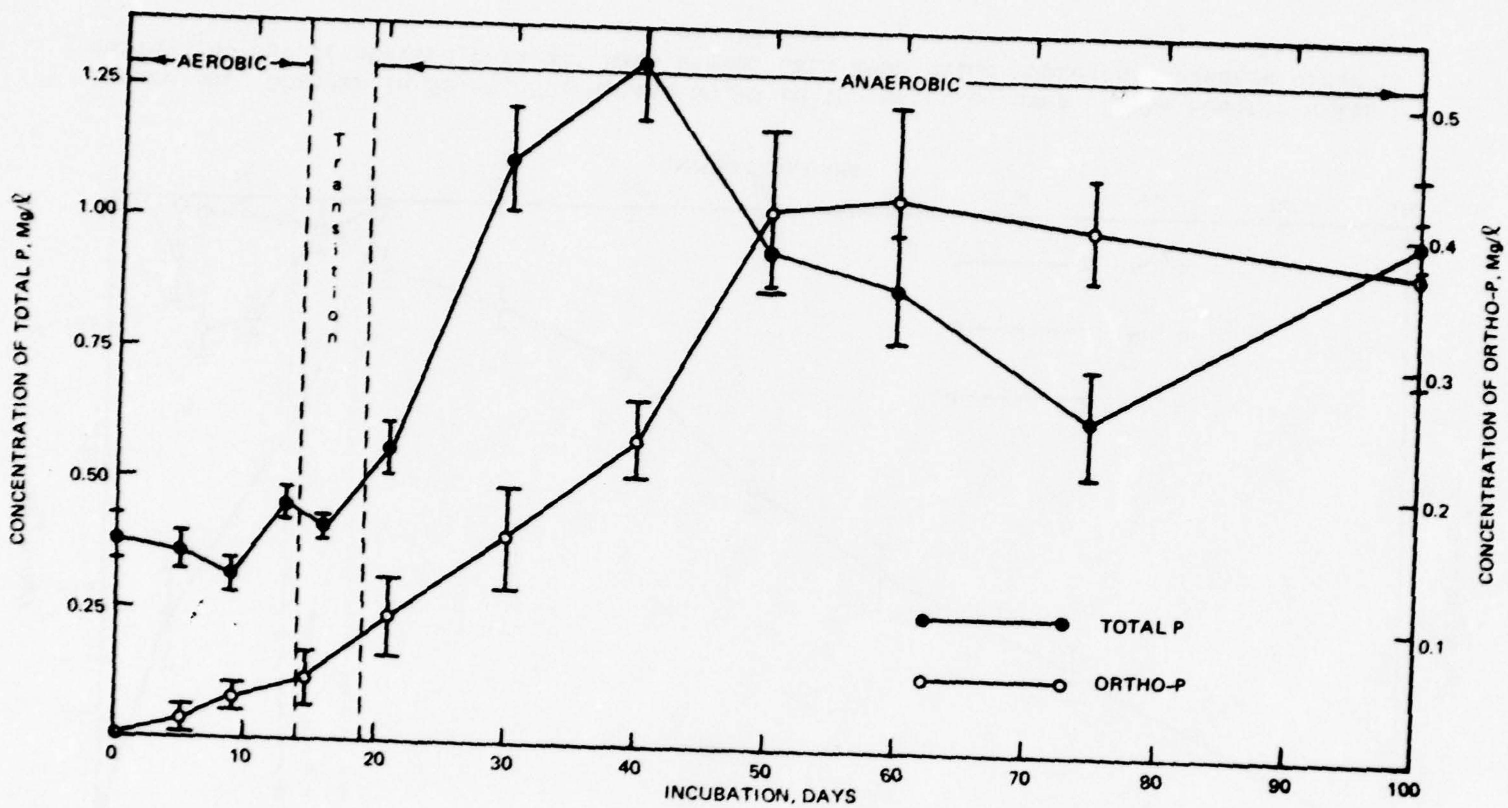

Figure B9. Changes in total P and ortho-P in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

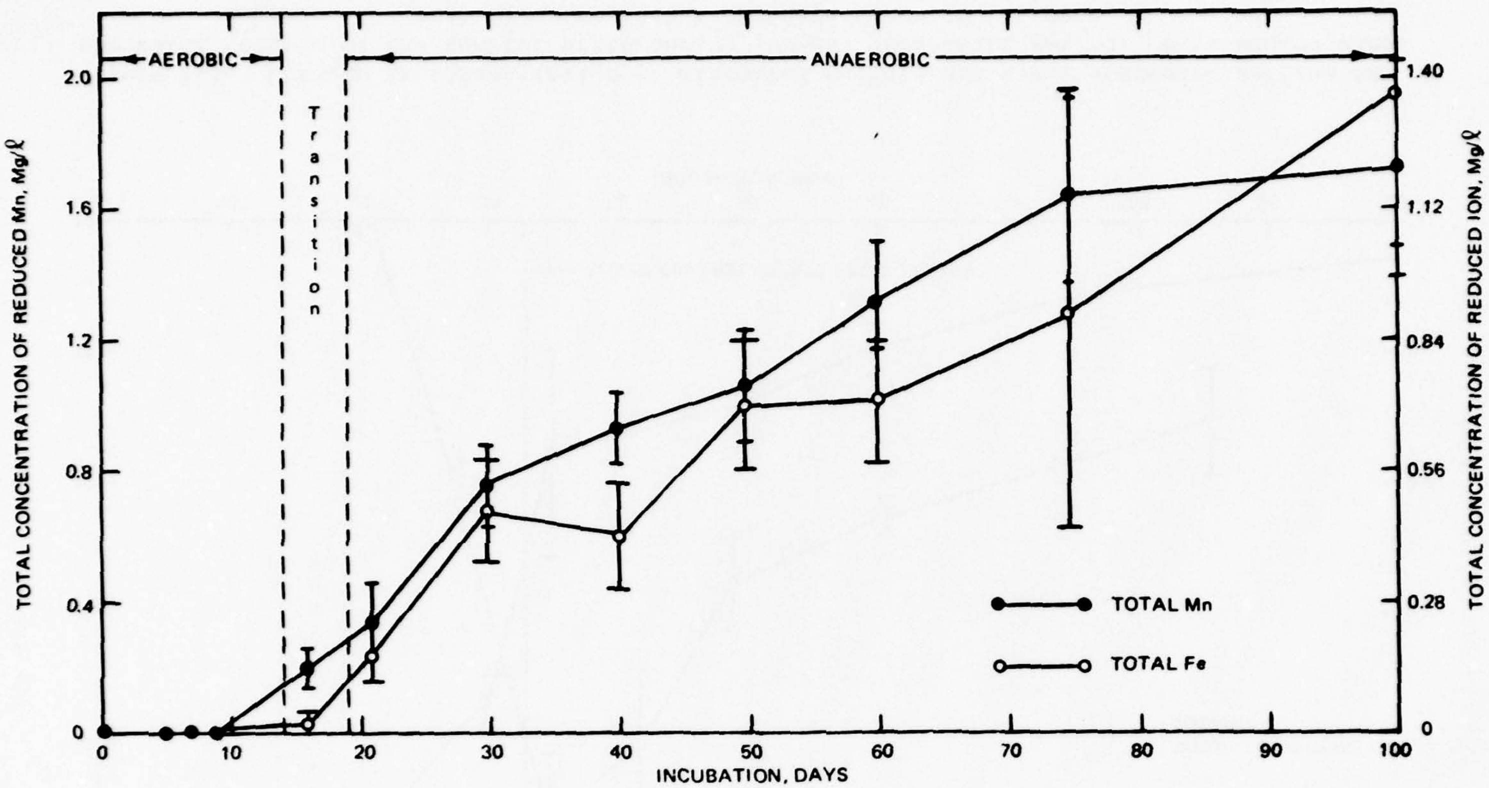

Figure B10. Changes in concentration of soluble reduced manganese (total Mn) and iron (total Fe) in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

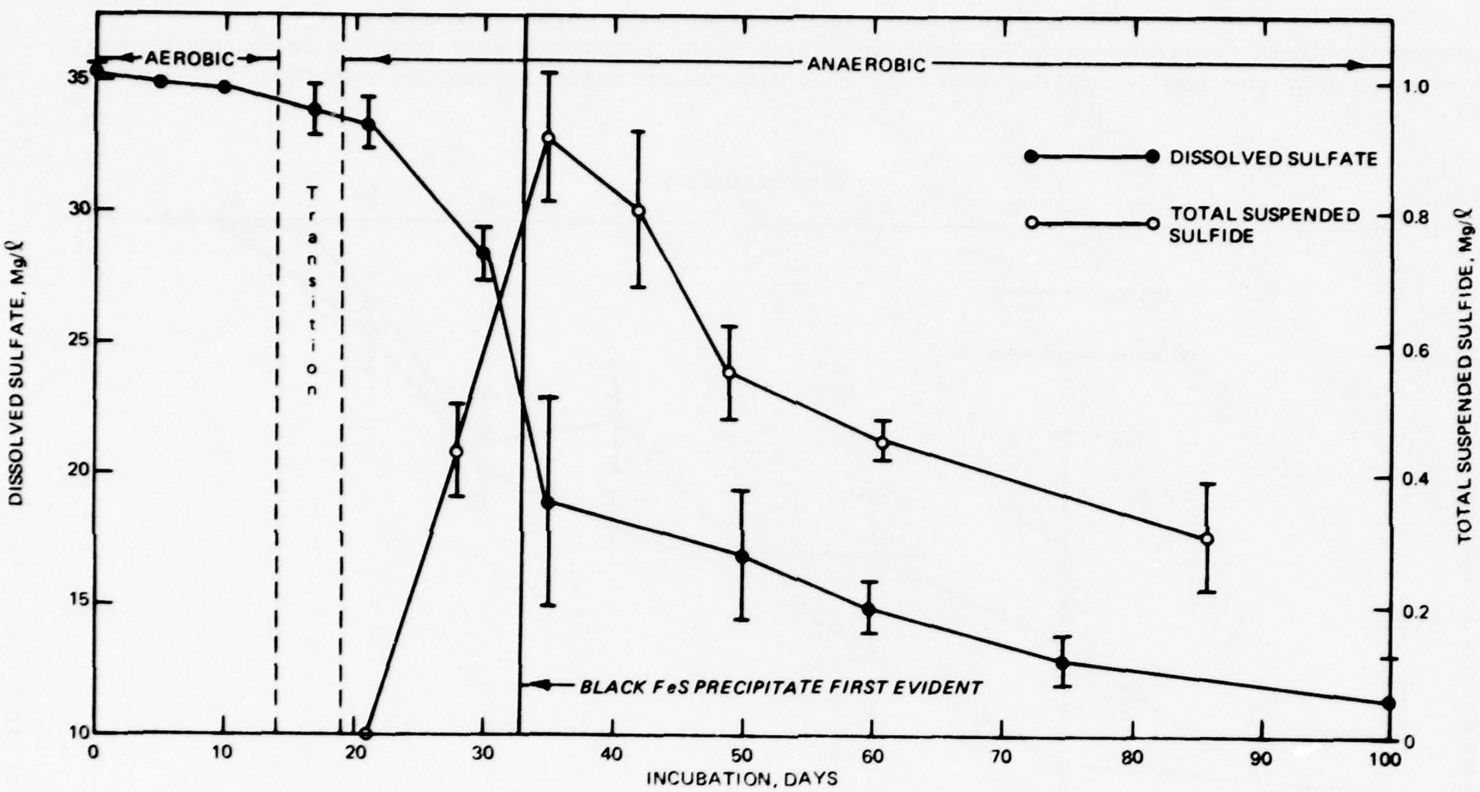

Figure B11. Changes in concentration of dissolved sulfate and total suspended sulfide in the water columns of the reactor units during 100-day incubation period. Bars around each mean value represent standard error of the mean

APPENDIX C:

PROJECTED ESTABLISHMENT AND GROWTH OF AQUATIC MACROPHYTES IN PROPOSED TWIN VALLEY LAKE

by

John W. Barko
Environmental Laboratory
U. S. Army Engineer Waterways Experiment Station

1. The presence of particular macrophyte species at a site depends on success of dispersal and suitability of the environment. Downstream dispersal by vegetative propagation and by seed is common in rivers. Thus, any macrophyte species upriver from the proposed Twin Valley impoundment are candidate colonizers. Other methods of dispersal (e.g., transport by waterfowl) may promote the colonization of macrophytes not presently existing in the watershed. Thus, most macrophyte species in temperate North America could potentially colonize the impoundment. However, on the basis of probability of propagation, the macrophyte community will most likely be comprised of species presently in the area. A listing of these species can be found in the St. Paul District final Environmental Impact Statement.

2. Because of changes in the relative suitability of the environment to different species, community composition will undergo continuous and unpredictable alteration associated with postimpoundment oscillations in the aquatic environment. Species equilibrium within the community will not likely be achieved for many years. During this period, the success of establishment of any particular species will be stochastically determined, but will be highly dependent upon specific dispersal ability.

3. Once a species is able to reproduce at a rate exceeding its rate of mortality, it can be considered successfully established. In uncolonized, yet suitable, environments plant growth rate is very rapid following establishment. As species begin to compete for environmental resources, they become sorted along specific environmental gradients. The evolutionary development of different life forms of aquatic plants has eased competition among species and thus maximized their utilization of environmental resources. The life forms of aquatic plants are given below with specific examples:

Life Form	Example
emergent	cattail
floating leaved	waterlily
submergent	pondweed
free floating	duckweed

4. Members of the first three categories (emergent, floating leaved, and submergent plants) are rooted and attached to a substratum. Free-floating plants often have roots, but these do not function in attachment. Emergent aquatic plants, although adapted to constant or periodic standing water, are most similar to terrestrial plants. These have easily distinguishable and functionally separable aerial and rooted portions. Submergent plants are the most highly adapted to the aquatic environment. These plants remain totally submersed throughout their lifespan, lack structural rigidity, and therefore are dependent upon the buoyancy of the water for their support. Floating-leaved plants are intermediate in form between emergent and submergent life forms and have characteristics of both. The existence of emergent, submergent, and free-floating life forms is likely in Twin Valley Lake. However, fluctuations in water level may impede the establishment of floating-leaved plants, which are more sensitive to changes in water depth during establishment than the other two life forms.

5. Once established, the growth of aquatic plants becomes dependent mainly on light and sediment, although temperature is also important. The relative importance of these factors varies with the environment; however, light penetration into the water is usually considered to be the factor limiting the depth distribution of submergent plants. In Twin Valley Lake, light is predicted to become limiting at a depth of approximately 3.3 m. In Figure C1 potentially colonizable lake surface area on the basis of light penetration alone is indicated. This area represents approximately 46 percent of the lake surface.

6. Emergent vegetation is obviously unaffected by light (in terms of limitation), but normally will not grow in water depths exceeding approximately 1 m. Thus, emergent plants in Twin Valley Lake will be restricted to immediate shoreline regions.

7. Since the majority of macrophytes are rooted, their growth and distribution are markedly affected by sediment type. The role of sediments in the nutrition of macrophytes is important because nutrients can be absorbed from the sediment by the roots. For the most part, regions of nutrient absorption depend upon the relative availability of

nutrients with respect to the distribution of plant tissues having an absorptive capability. Nutrient uptake by emergent macrophytes, which have extensive root systems and little absorptive surface exposed to the water, probably occurs almost exclusively from the sediment. In the case of submergent and floating-leaved macrophytes, nutrients may be absorbed from either the sediment or the water. However, recent evidence has suggested that the roots of these life forms are functionally similar to those of emergent macrophytes. The nutrition of these life forms is probably much more commonly sediment based because of the greater availability of nutrients in sediments. Predicted high sediment loads into Twin Valley should provide an ideal nutritional environment for rooted macrophytes. Likewise, however, sediment-associated turbidity with consequent reduced light penetration may locally impede the distribution of submergent plants.

8. The influence of high macrophyte density on water quality could be important because of the extensiveness of the littoral zone in the proposed impoundment. For example, recent evidence indicates that rooted plants may significantly affect pelagic and sedimentary nutrient cycles by returning to the water column nutrients that would otherwise remain sorbed to sediment particles. Releases of nutrients from macrophytes are dependent upon tissue sloughing and decay. These processes occur throughout the growing season and may provide sustained nutrient inputs supporting phytoplankton productivity during periods of reduced inflow, when nutrients might otherwise limit algal growth.

9. The primary effect of water temperature on macrophytes is on species composition because different plants often demonstrate distinct temperature optima. Indirect effects of temperature are complex, but nonetheless important. For example, the solubility of gases and rates of microbial mineralization of organic matter are temperature-dependent processes that can affect both plant growth and the influence of plants on nutrient cycles. Water temperatures in Twin Valley Lake will be within a range supportive of most macrophyte species in temperate regions of the country.

Figure C1. Proposed Twin Valley Lake macrophyte areas

APPENDIX D: INITIAL CONDITIONS, COEFFICIENTS, AND UPDATES FOR MATHEMATICAL ECOLOGICAL SIMULATIONS

Table D1

Summary of Initial Conditions for Ecological Simulations

Parameter	1971	1975	1976
Julian day	100	109	87
Pool elevation, m	9.5	10.6	9.5
Fish			
Predators (FISH1),* kg/ha	10	10	10
Planktivores (FISH2), kg/ha	20	20	20
Benthos feeders (FISH3), kg/ha	44	44	44
Water quality parameters			
Algae 1 (ALGAE1), mg/ℓ	1.0	1.0	1.0
Algae 2 (ALGAE2), mg/ℓ	0.1	0.1	0.1
Alkalinity (ALK), mg/ℓ	190	192	220
Benthos (BEN), mg/m^2	1650	1650	1650
Ammonia (NH3), mg/ℓ N	0.04	0.07	0.20
Nitrite (NO2), mg/ℓ N	0.0	0.0	0.0
Nitrate (NO3), mg/ℓ N	0.10	0.54	0.34
Fecal coliforms (COL), colonies/100 ml	58	58	20
Detritus (DET), mg/ℓ	3.4	3.4	3.4
Dissolved organics (DOR), mg/ℓ	5.5	5.5	4.6
Dissolved oxygen (DO), mg/ℓ	13.4	13.0	13.2
Orthophosphate (PO4), mg/ℓ P	0.04	0.04	0.05
Organic sediment (SED), mg/m^2	5600	5600	5600
Temperature (TEMP), ^oC	3.4	4.1	3.6
Total dissolved solids (TDS), mg/ℓ	264	283	284
Zooplankton (ZOO), mg/ℓ	0.2	0.2	0.2
pH (PH)	8.2	8.1	7.9

* Acronym in parenthesis represents the variable name used in the U. S. Army Engineer Waterways Experiment Station (WES) version of the Water Quality for River-Reservoir Systems model.

Table D2

Coefficients for Base Simulation

Parameter	Coefficient
Physical coefficients	
Turbidity factor (TURB)*	2
Evaporative wind function (AA+BB*WIND)	
AA	0 m/(sec-mb)
BB	1.2×10^{-9} mb^{-1}
Mixing coefficients	
Stability parameter (GSWH)	7.0×10^{-5} sec^{-2}
Wind mixing coefficient (A1)	3.0×10^{-4}
Hypolimnetic diffusivity (A2)	5.0×10^{-6} m^2/sec
Metalimnetic coefficient (A3)	-0.4
Extinction coefficient (EXCO)	0.7 m^{-1}
Surface radiation fraction (SURFRAC)	0.5
Critical advective density (CDENS)	2.0 kg/m^3
Reaeration coefficients	
Oxygen (DMO2)	2.04×10^{-9} m^2/sec
Carbon dioxide (DMCO2)	2.04×10^{-10} m^2/sec
Stoichiometry	
02 - NH3 (02NH3)	3.5
02 - NO2 (02NO2)	1.2
02 - Detritus (02DET)	2.0
02 - Respiration (02RESP)	1.6
02 - Algal biomass (02FAC)	1.6
CO2 - Dissolved organics (CO2DOR)	0.2
Decay rates	
Dissolved organics (TDORDK)	0.15 per day
Ammonia (TNH3DK)	0.18 per day
Nitrite (TNO2DK)	0.40 per day

(Continued)

* Acronym in parenthesis represents the variable name used in the WES version of the Water Quality for River-Reservoir Systems model.

(Sheet 1 of 5)

Table D2 (Continued)

Parameter	Coefficient
Decay rates (cont'd)	
Coliforms	
(Q10)	1.04
(TCOLDK)	1.4 per day

Parameter	Coefficient ALGAE 1 I = 1	Coefficient ALGAE 2 I = 2
Algae		
Chemical composition		
C	0.45	0.45
N	0.08	0.08
P	0.011	0.011
Gross production rate (TPMAX(I))	1.42 per day	1.61 per day
Temperature rate multipliers		
Lower threshold (T1)	$0^{o}C$	$5^{o}C$
Optimum (T2)	$18^{o}C$	$20^{o}C$
Optimum (T3)	$22^{o}C$	$27^{o}C$
Upper threshold (T4)	$28^{o}C$	$36^{o}C$
Half-saturation coefficients		
Carbon (PS2CO2(I))	0.10 mg/ℓ	0.10 mg/ℓ
Nitrogen (PS2N(I))	0.014 mg/ℓ	0.010 mg/ℓ
Phosphorus (PS2PO4(I))	0.003 mg/ℓ	0.006 mg/ℓ
Light (PS2L(I))	6.0 kcal/m^2/hr	4.0 kcal/m^2/hr
Respiration rate (TPRESP)	0.17 per day	0.17 per day
Settling rate (TSETL(I))	0.3 m/day	0.1 m/day
Self-shading coefficient	0.1 per m-mg/ℓ	0.1 per m-mg/ℓ

(Continued)

(Sheet 2 of 5)

Table D2 (Continued)

Parameter	Coefficient
Zooplankton	
Chemical composition	
C	0.45
N	0.08
P	0.012
Assimilation rate (TZMAX)	0.505 per day
Temperature rate multipliers	
Lower threshold (T1)	$0^{o}C$
Optimum (T2)	$20^{o}C$
Optimum (T3)	$26^{o}C$
Upper threshold (T4)	$36^{o}C$
Assimilation efficiency (ZEFFIC)	0.65
Feeding preference	
Algae 1 (PREF(1))	0.7
Algae 2 (PREF(2))	0.3
Detritus (PREF(3))	0
Half-saturation coefficient (ZS2P)	0.2 mg/ℓ
Mortality rate (TZMORT)	0.005 per day
Respiration rate (TZRESP)	0.2 per day
Detritus	
Chemical composition	
C	0.32
N	0.07
P	0.009
Settling rate (TTSETL)	0.15 m/day
Decay rate (TDETDK)	0.09 per day
Benthos	
Chemical composition	
C	0.47
N	0.08
P	0.011

(Continued)

(Sheet 3 of 5)

Table D2 (Continued)

Parameter	Coefficient
Benthos (cont'd)	
Assimilation rate (TBMAX)	0.04 per day
Temperature rate multipliers	
Lower threshold (T1)	$0^{o}C$
Optimum (T2)	$20^{o}C$
Optimum (T3)	$26^{o}C$
Upper threshold (T4)	36^{o}
Assimilation efficiency (BEFFIC)	0.6
Half-saturation coefficient (BS2SED)	200 mg/m^2
Mortality rate (TBMORT)	0.005 per day
Respiration rate (TBRESP)	0.016 per day

Parameter	Coefficient FISH 1 I = 1	Coefficient FISH 2 I = 2	Coefficient FISH 3 I = 3
Fish			
Chemical composition			
C	0.45	0.45	0.45
N	0.08	0.08	0.08
P	0.011	0.011	0.011
Assimilation rate (TFMAX(I))	0.013 per day	0.014 per day	0.014 per day
Temperature rate multipliers			
Lower threshold (T1)	$0^{o}C$	$0^{o}C$	$0^{o}C$
Optimum (T2)	$25^{o}C$	$25^{o}C$	$25^{o}C$
Optimum (T3)	$29^{o}C$	$29^{o}C$	$29^{o}C$
Upper threshold (T4)	$35^{o}C$	$35^{o}C$	$35^{o}C$
Assimilation efficiency (FEFFIC)	0.8	0.8	0.8

(Continued)

(Sheet 4 of 5)

Table D2 (Concluded)

Parameter	Coefficient FISH 1 I = 1	FISH 2 I = 2	FISH 3 I = 3
Fish (cont'd)			
Half-saturation coefficients			
Fish (FS2FSH)	5.7 kg/ha	--	--
Zooplankton - Detritus (FS2ZOO)	--	0.76 mg/ℓ	--
Benthos - sediment (FS2BEN)	--	--	7.0 mg/ℓ
Fraction of diet			
Sediment (F3SED)	--	--	0.45
Benthos (F3BEN)	--	--	0.55
Mortality rate (TFMAX)	0.002 per day	0.002 per day	0.002 per day
Respiration rate (TFRESP)	0.008 per day	0.008 per day	0.008 per day

(Sheet 5 of 5)

Table D3

1971 Water Quality Updates

Parameter	Update	
ALG 1, mg/ℓ	Assume = 0.37	
ALG 2, mg/ℓ	Assume = 0	
ALK, mg/ℓ	$ALK(I) = 246(FLOW(I))^{-0.076}$	$R^2 = 0.51$
BOD, mg/ℓ	Assume = 5.5	
NH4-N, mg/ℓ	Assume = 0.04	
NO2-N, mg/ℓ	Assume = 0.0	
NO3-N, mg/ℓ	$NO3N(I) = 0.0251(FLOW(I))^{0.41}$	$R^2 = 0.80$
COL, colonies/100 ml	Assume = 58	
DET-C, mg/ℓ	Assume = 3.4	
DO, mg/ℓ	Assume 94 percent saturation	
PO4-P, mg/ℓ	$PO4P(I) = 0.0235(FLOW(I))^{0.14}$	$R^2 = 0.92$
TEMP, ^oC	Generated from air temperature residual	
TDS, mg/ℓ	$TDS(I) = 316(FLOW(I))^{-0.055}$	$R^2 = 0.50$
ZOO, mg/ℓ	Assume = 0	
PH	Assume = 8.2	

Table D4

1975 Water Quality Updates

Julian Day	ALG 1	ALG 2	ALK	BOD	NH4-N	NO2-N	NO3-N	COL	DET	DO	PO4-P*	TEMP	TDS	ZOO	PH
69			311		0.33		0.14				0.01		353		7.3
119			162		0.00		0.64				0.05		266		8.3
154			218		0.02		0.01				0.02		279		7.9
184	Assume = 0.37	Assume = 0	171	Assume = 5.5	0.04	Assume = 0	0.13	Assume = 58	Assume = 3.4	Assume 94 percent saturation	0.09	Generated from air temperature residual	283	Assume = 0	7.8
196			207		0.00		0.06				0.07		286		8.2
238			229		0.00		0.02				0.03		297		8.5
280			296		--		--				0.02		360		8.1

* PO4-P = 0.5*P.

Note: All concentrations are in mg/ℓ. Fecal coliforms (COL) are in colonies/100 ml and temperature is in °C.

Table D5

1976 Water Quality Updates

Julian Day	ALG 1	ALG 2	ALK	BOD	NH4-N	NO2-N	NO3-N	COL	DET	DO	PO4-P	TEMP	TDS	ZOO	PH
75			302	--	0.23		0.26	16			--		347		7.3
97			150	4.6	0.18		0.41	22			0.05		231		8.4
104			176	4.2	0.05		0.16	13			0.01		244		8.5
112			197	2.8	0.02		0.02	25			0.02		247		8.4
117			212	3.7	0.01		0.00	16			0.00		264		8.5
124	Assume = 0.37	Assume = 0	222	2.4	0.01	Assume = 0	0.01	20	Assume = 3.4	Assume 94 percent saturation	0.00	Daily data	264	Assume = 0	8.3
133			238	--	0.02		0.01	7			0.00		283		8.6
145			240	1.5	0.02		0.04	24			0.01		271		7.9
154			237	1.0	0.02		0.01	--			0.01		342		8.4
159			241	1.6	0.01		0.02	44			0.01		299		8.3
166			233	1.9	0.04		0.01	120			0.01		275		8.5
173			220	1.4	0.03		0.04	390			0.02		267		8.3
180			225	5.5	0.02		0.03	148			0.01		272		7.8
189			217	2.6	0.01		0.01	68			0.02		281		8.3
194			221	1.3	0.07		0.01	67			0.04		276		8.5
208			245	1.4	0.00		0.00	22			0.08		317		8.5
215			267	--	0.00		0.00	40			0.03		343		7.5

(Continued)

Note: All concentrations are in mg/ℓ. Fecal coliforms (COL) are in colonies/100 ml and temperature is in ^{o}C.

Table D5 (Concluded)

Julian Day	ALG 1	ALG 2	ALK	BOD	NH4-N	NO2-N	NO3-N	COL	DET	DO	PO4-P	TEMP	TDS	ZOO	PH
222	Assume = 0.37	Assume = 0	264	--	0.00	Assume = 0	0.01	26	Assume = 3.4	Assume 94 percent saturation	0.02	Daily data	353	Assume = 0	8.6
229			253	--	0.00		0.01	39			0.00		299		8.5
237			224	--	--		0.39	71			0.03		357		8.2
243			271	--	0.02		0.01	55			0.04		333		8.5
251			276	1.8	0.01		0.01	68			0.03		361		8.3
258			268	1.4	0.04		0.00	76			0.04		361		7.5
264			274	6.4	0.03		0.01	8			0.04		351		8.3
272			289	1.8	0.01		0.01	156			0.02		384		8.5
279			291	--	0.00		0.00	--			0.01		375		8.2
286			314	--	0.00		0.00	--			0.03		393		7.4
293			323	--	0.0		0.01	--			0.02		438		8.3
302			347	--	0.01		0.01	--			0.03		435		8.5

APPENDIX E: COEFFICIENT REFERENCES

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WATER QUALITY EVALUATION OF PROPOSED TWIN VALLEY LAKE, WILD RIC--ETC(U)
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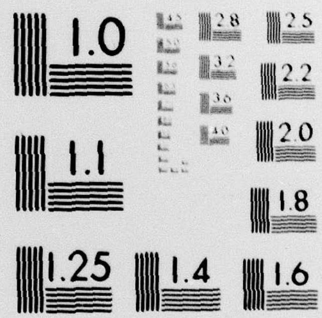

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APPENDIX F:

NUTRIENT LOADING CALCULATIONS

Table F1

Nutrient Loading Models and Phosphorus Retention Equation

Source*	Equation
Dillon and Rigler (1974)	$\{P\} = \frac{L\ (1 - R)}{\bar{z}\rho}$
Kirchner and Dillon (1975)	$R = 0.426 \exp(-0.271\ q_s) + 0.574 \exp(-0.00949\ q_s)$
Larsen and Mercier (1976)	$\{P\} = \{\bar{p}\}\ (1 - Rp)$
Larsen and Mercier (1976)	$Rp = \frac{1}{1 + \rho^{1/2}}$
Vollenweider (1976)	$L_c = (10\text{-}20)\ q_s\ (1 + \sqrt{\bar{z}/q_s})$
Vollenweider (1976)	$\{P\} = \frac{L}{qs\left\{1 + \left(\frac{\bar{z}}{q_s}\right)\right\}}$

* See references at end of main text.

NOTE: $\{P\}$ = average phosphorus concentration, mg/m^3

L = annual phosphorus loading, $g/(m^2 \times yr)$

R, Rp = retention coefficients

$\bar{z}$ = mean depth, m

ρ = flushing rate per year, yr^{-1}

q_s = areal water load or Q/A

Q = annual outflow, m^3/yr

A = Surface area of impoundments, m^2

$\{\bar{p}\}$ = average influent P concentration

L_c = critical annual phosphorus loading, $g/(m^2 \times yr)$

Table F2

Predicted Average In-Lake Phosphorus Concentrations, mg/ℓ, Based on Regression Equation vs. Mean Concentration

Method	1971		1975		1976	
	Mass	Conc	Mass	Conc	Mass	Conc
Vollenweider (1976)	52	46	64	49	49	43
Larsen & Mercier (1976)	52	46	64	49	49	43
Dillon & Rigler (1974)	47	42	65	49	40	35

Table F3

Trophic State Index (TSI) Calculated from Predicted In-Lake Phosphorus Concentrations (Carlson 1977)

Phosphorus Equation Source	1971		1975		1976	
	Mass	Conc	Mass	Conc	Mass	Conc
Vollenweider (1976)	61	59	64	60	60	58
Larsen & Mercier (1976)	61	59	64	60	60	58
Dillon & Rigler (1974)	60	58	64	60	57	55

Table F4

Equations Used to Predict Chlorophyll Concentrations

	Chlorophyll Analyses
Vollenweider (1976)	$[\overline{chla}] = 0.367 [P]^{0.91}$
Dillon & Rigler (1974)	$\log_{10} [chla] = 1.45 \log_{10} [P] - 1.14$
Carlson (1977)	$\ln [chla] = 1.449 \ln [P] - 2.442$ Jul-Aug

Note: chla = chlorophyll a concentration, μg/ℓ

P = average phosphorus concentration, mg/m^3.

Table F5

Predicted Chlorophyll Concentrations (μg/ℓ) for Proposed Twin Valley Lake Based on Estimated Phosphorus Concentrations

Equation	1971		1975		1976	
	Mass	Conc	Mass	Conc	Mass	Conc
Vollenweider (1976)	13	12	16	13	13	11
Dillon & Rigler (1974)	19	16	31	20	15	12
Carlson (1977)	25	21	36	24	22	18

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Ford, Dennis E
  Water quality evaluation of proposed Twin Valley Lake, Wild Rice River, Minnesota / by Dennis E. Ford ... [et al.]. Vicksburg, Miss. : U. S. Waterways Experiment Station ; Springfield, Va. : available from National Technical Information Service, 1979.
  66, [228] p. : ill. ; 27 cm. (Technical report - U. S. Army Engineer Waterways Experiment Station ; EL-79-5)
  Prepared for U. S. Army Engineer District, St. Paul, St. Paul, Minnesota.
  Includes bibliographies.

1. Bioassays. 2. Ecological models. 3. Eutrophication. 4. Mathematical models. 5. Nutrient loadings. 6. Twin Valley Lake. 7. Water quality. 8. Wild Rice River. I. United States. Army. Corps of Engineers. St. Paul District. II. Series: United States. Waterways Experiment Station, Vicksburg, Miss. Technical report ; EL-79-5.
TA7.W34 no.EL-79-5